

## **APPENDIX B**

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### **An Assessment of Potential Risks Posed by PAHs, PCBs, and DDT in Dredged Material to Juvenile Salmonids in the Lower Columbia River: Mouth to Bonneville Dam**

## 1 EXECUTIVE SUMMARY

The objective of this report is to determine whether improvements to the Columbia River navigation channel will exacerbate the risks posed by bioaccumulative contaminants to juvenile salmonids that rear in the lower Columbia River and feed on epibenthic invertebrates. The report also examines potential risks to these prey. The contaminants assessed include compounds that are environmentally persistent and bioaccumulate in fish and invertebrates, namely total polychlorinated biphenyls ( $\Sigma$ PCBs); DDT and its metabolites, DDD and DDE ( $\Sigma$ DDT); and total polycyclic aromatic hydrocarbons ( $\Sigma$ PAHs). Preliminary evidence obtained by the National Marine Fisheries Service (NMFS) suggests that some endangered salmonid stocks may be at risk from the effects of the contaminants contained in the tissues of their epibenthic prey. The origin of these contaminants is unclear and could even be from exposures incurred upstream before the fish reach the estuary. However, it is generally assumed that some types of dredging suspend fine particulates. It has been hypothesized that if these types of particulates are suspended by dredging operations in the lower Columbia River the particulates may be entrained within the estuarine turbidity maximum (ETM) zone. The ETM is an area where epibenthic prey of juvenile salmon are assumed to thrive. Therefore, there may be a potential risk to juvenile salmonids as well as their prey, and the purpose of this assessment is to examine it.

A risk-based approach was used to address possible effects of contaminants on salmonids in the Columbia River Basin. Risks were evaluated by comparing the frequency and magnitude of contaminant exposures in sediments with aquatic toxicological data to predict the frequency and magnitude of adverse effects expected from sediment exposures. Data defining contaminant exposures in Columbia River sediments from the mouth to the head of the tidewater near Bonneville Dam were obtained from the majority of available data sources. These included a regional sediment database (SEDQUAL) maintained by the Washington State Department of Ecology, data obtained by the U.S. Army Corps of Engineers, and data obtained by Tetra Tech, Inc., in a "Bi-State Survey" of sediment contamination in the lower Columbia River. Exposures were defined for three reaches of the Columbia estuary: River Mile (RM) 0-40, RM 41-101 (the confluence with the Willamette River), and RM > 101 (from the Willamette confluence to the head of the tidewater). Data defining the range of potential effects were obtained from sediment screening guidelines developed by regional state and federal agencies, from scientific literature, and from technical reports and data analyses conducted by scientists with NMFS. Risks were defined by comparing the probabilities of exposure with those of effects. Many of the effects occurring at the lowest contaminant concentrations are sublethal, indirect responses, such as histopathological changes, changes in enzyme activity, decreased growth, or decreased disease resistance. For the purposes of this report, it has been assumed that such responses result in mortality although direct evidence linking sublethal effects to increased mortality or decreased reproductive success has not been demonstrated anywhere at the low contaminant concentrations typical of sediments in the Columbia estuary.

Figures B-1 through B-3 summarize the results of the analyses for juvenile salmonids exposed in the lower Columbia estuary (RM 0-40) to all three contaminant classes. As indicated in all three figures, only negligible risks were predicted for the channel sediments proposed for dredging. Risks were present in some of the sediment that had been sampled nearshore, outside of the shipping channel, but they appeared localized to certain sources and sediment types. All  $\Sigma$ PCB exposures in the shipping channel were just below the effects threshold (10% effects) proposed by a NMFS scientist (Meador 2000a), and 10 times below the regional sediment screening guideline (Figure B-1). Consequently,  $\Sigma$ PCB risks in the channel should be negligible. Likewise, all  $\Sigma$ DDT exposures via channel sediments were below both the regional screening guideline and a lowest observed effect threshold developed from testing of cutthroat trout (Figure B-2). Cutthroat trout appear to be the salmonid most sensitive to DDT, so these results may be applied as a conservative standard to other juvenile salmonids. Finally, all  $\Sigma$ PAH exposures via channel

sediments were lower than four criteria establishing potential effects. For example, channel sediment PAH concentrations were 41 parts per billion (ppb) dry weight (dw) or lower, whereas the lowest potential effect criterion proposed by Johnson (2000) was 54 ppb dw. Other effect criteria were much higher, ranging from 1,000 to 15,100 ppb dw (Figure B-3).

Risks to the sediment-dwelling prey of juvenile salmon in the shipping channel were also below all effect thresholds. Some of the sediments nearshore, outside of the shipping channel, posed risks. The magnitudes and character of the risks basically were the same from the lower estuary to Bonneville Dam.

The main reason why risks were negligible to juvenile salmon and their prey is because most lower Columbia River channel sediments are essentially devoid of organic carbon, the substrate to which the persistent, bioaccumulative contaminants adsorb. The microbial biofilm that accumulates on the surface of organic particles is the food of epibenthic invertebrates and the apparent pathway by which these contaminants enter food chains involving juvenile salmon. The channel sediments, which are almost exclusively sand, are largely devoid of organic carbon because channel currents sweep away most of the fine particulates. As a consequence, concentrations of contaminants, if present at all, are actually lower than those in nearshore (depositional) environments. Accordingly, if resuspension of sediments during dredging did occur, it is not expected to change the concentration of contaminants found in nearshore environments.

The potential for cumulative risks from exposure to all three contaminant classes is negligible because none of the classes exceeded effects thresholds and these risks do not appear to be additive or more-than-additive (synergistic). Because the specific modes of action of all PCB and PAH compounds considered, as well as those of DDT, DDD, and DDE, are believed to be different, and because exposures were below effects thresholds, risks from the different classes of compounds should not be added. The overall conclusion is one of negligible risk to juvenile salmonids and their sediment-dwelling prey in the lower Columbia River as a result of the dredging associated with the lower Columbia River navigation channel deepening project.

## 2 INTRODUCTION

### 2.1 Background

The Portland District of the U.S. Army Corps of Engineers (Corps,) has proposed to deepen the lower Columbia River navigation channel and has prepared an environmental impact statement (EIS) on this proposal (Corps, 1998). In its biological opinion concerning the potential effects of proposed channel improvements on endangered salmonid stocks, the National Marine Fisheries Service (NMFS, 1999) questioned whether dredging associated with channel deepening would pose risks to juvenile salmonids in the lower Columbia River by enhancing the availability and toxicity of certain chemicals in their prey. It specifically requested assessment of "...potential impact of contaminants from redistribution by dredging activities, incorporating appropriate endpoints to derive realistic sediment quality standards and utilizing bioaccumulation potential from salmon-associated prey base as an additional source of transfer of contaminants to juvenile salmon."<sup>1</sup>

This study focused specifically on the lower Columbia River and estuary. The estuary is especially important in the life cycle of juvenile salmon and steelhead trout because that is where they delay their seaward migration, usually for days to weeks, to feed and adapt to seawater (Aitkin, 1998). In his review, Casillas (1999) identified two studies that documented significantly improved smolt-to-adult survival of salmon from residence in the Columbia River estuary. Specific attention is paid to ocean-type juvenile chinook because they – along with chum salmon – appear to be most dependent on the lower estuary for rearing.<sup>2</sup> In addition, their diet tends to consist predominantly of the invertebrates that live on or at the sediment's surface (epibenthic) until the salmon grow large enough to feed pelagically in deeper waters (Aitkin, 1998).

Chemicals with certain properties can be toxic when consumed in the diet of fish. For at least the past 20 years, the literature has shown that chemicals with certain properties – namely hydrophobicity and resistance to metabolism and excretion (Macek, et al., 1979) – will tend to attach to fine particulate organic matter, be bioaccumulated, and persist in the food web of fish and wildlife (Lake, et al., 1987). Some of the sediment-dwelling (benthic) prey of juvenile salmon, such as the amphipod *Corophium salmonis*, are known to feed in part on organic matter (detritus). Thus, because some types of dredging may suspend the detritus, dredging could potentially make the detritus – and any contaminants bound to it – more accessible to benthic organisms, as shown in Figure B-4. Chemicals having the highest potential risk because of these properties (i.e., hydrophobicity and resistance to metabolism and excretion) are the highly chlorinated hydrocarbons (PCBs<sup>3</sup>) and insecticides (DDT<sup>4</sup>). The risks posed by these chemicals via food chain (i.e., dietary) exposure of fish-eating birds are well known (Hoffman, et al., 1990). Until the mid-1980s, most studies examining the risks posed to fish have suggested that these chemicals were bioaccumulated primarily via uptake across the gills (e.g., Adams, et al., 1985; Dobroski and Epifanio, 1980; Macek, et al., 1979). The water and dietary exposure pathways, both of which are now known to contribute variably to bioaccumulation (Di Toro, et al., 1991; EPA, 2000), are depicted in Figure B-5. In

<sup>1</sup> Here, salmon (*Oncorhynchus*), trout (*Salmo*), and char (*Salvelinus*) are referred to as salmonids because they are in the same family, Salmonidae.

<sup>2</sup> Ocean-type chinook spend less than 1 year and as little as 3 months in freshwater before migrating seaward. They enter the estuary as small as 40 millimeter (mm) in length and usually leave when they reach 70 mm or larger size (Aitkin, 1998). The larger chum salmon juveniles that reside in the channel should have switched from feeding on epibenthic to feeding on primarily pelagic invertebrates, and thus should incur limited exposure to epibenthic prey in the channels.

<sup>3</sup> Polychlorinated biphenyls are a large group of chlorinated hydrocarbons constituting dozens of compounds.

<sup>4</sup> Chemical Name for DDT: 1,1'-(2,2,2-Trichloroethylidene)bis(4-chlorobenzene) (CAS # 50-29-3).

the Columbia River estuary, the relationships between the dietary pathway and the food webs pertinent to juvenile salmon and trout (salmonids) are shown in Figure B-6.

In addition, other chemicals have been identified as being of special concern to NMFS. Research by NMFS and others in the Pacific Northwest have confirmed that polycyclic aromatic hydrocarbons (PAHs) and the antifouling biocide, tributyltin (TBT), behave similarly to chlorinated hydrocarbons and insecticides in terms of sorption to organic carbon and dietary accumulation (McCain, et al., 1990; Meador, et al., 1995; Meador, 2000a). Moreover, work by NMFS scientists has documented that concentrations of PAHs, PCBs, and DDE<sup>5</sup> are elevated in the sediments of urban Puget Sound estuaries and that juvenile salmon may be ingesting elevated residues of these substances when they feed (Johnson, 2000; McCain, 1990; Stein, et al., 1995; Varanasi, et al., 1993). However, most PAHs are considered inherently less hazardous than PCBs and chlorinated insecticides because they are metabolized relatively rapidly (Niimi and Palazzo, 1986) and consequently do not biomagnify up the food web as the chlorinated insecticides and PCBs do (Suedel, et al., 1994). Nevertheless, this does not imply that PAHs are unimportant environmental contaminants. Elevated sediment PAH concentrations have been associated with many pathologies in aquatic organisms, especially species like brown bullhead (*Ictalurus punctatus*), English sole (*Parophrys vetulus*), and amphipods, which live near or on the sediment (e.g., see Balch, Metcalfe, and Huestis, 1995; Johnson, 2000; Myers et al. 1991; Swartz, 1999).

When salmonids ingest chemicals bioaccumulated in their prey, a variable fraction is bioaccumulated and the remainder is excreted with unassimilated organic carbon in the feces (Gobas, et al., 1989). The bioaccumulated residues can elicit toxic effects if their concentrations are high enough (Jarvinen and Ankley, 1999; Johnson, 2000; Meador, 2000). Above a certain concentration, called a threshold, the number and severity of toxic effects increases with the dose.

The effects of dredging activities have been studied extensively for several decades, and these studies have resulted in development of criteria defining the suitability of dredging and disposal of sediments based on their concentrations of toxic substances. In the Pacific Northwest, marine sediment quality criteria have been developed by a consortium of state and federal agencies under the Puget Sound Dredged Disposal Analysis Program (Corps, et al., 2000). The Washington State Department of Ecology (WSDOE) has developed sediment quality standards ([http://www.ecy.wa.gov/programs/tcp/smu/sed\\_chem.htm](http://www.ecy.wa.gov/programs/tcp/smu/sed_chem.htm)), but they pertain mainly to sediments discharged from point and non-point sources and hazardous waste sites.

Because dredging is a transient, temporary action, there has long been concern about possible acute sublethal effects of toxicants on juvenile salmon and steelhead, which are exposed only for brief periods in lower rivers and estuaries (Servizi, 1990). Concern about dredging effects has historically focused on dissolved oxygen, physical effects from turbidity (e.g., on fish gills), and fish's ability to see food and avoid predators. Recently, concern has shifted to effects resulting from subtle, sublethal responses. The NMFS (1999) biological opinion and many studies identified by NMFS (e.g., Varanasi, et al., 1993; Arkoosh, et al., 1998) re-emphasized concerns about adverse sublethal effects of contaminants in foods being consumed by salmon juveniles. Sublethal effects include a broad range of behavioral, biochemical, immune, oncological, physiological, and whole organismic responses to toxicant exposure. Sublethal effects should be evaluated to determine whether there is evidence that they result, directly or indirectly, in the mortality of the organism and, if so, whether mortalities affect enough individuals to constitute harm to each endangered stock of salmonid, i.e., evolutionarily significant unit (ESU).

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<sup>5</sup> DDE is a metabolite of DDT that is highly refractory to degradation.

Sublethal effects resulting from ingestion of bioaccumulative chemicals of concern in food (i.e., the dietary exposure pathway) are difficult to study. Data relating dose to response with respect to sublethal effects, or relating sublethal effects to those on the ESU population, are limited. Exposure pathways are especially difficult to study because the sublethal effects that arise from the variable and short durations (a few weeks to several months, averaging perhaps one month), that ocean-type juvenile chinook spend in estuaries (Aitkin, 1998). Most of the aquatic toxicological data reported in the scientific literature has focused on acute effects on survival and chronic effects on survival, growth, and reproduction rather than sublethal effects (see EPA's Ecotox database – [http://www.epa.gov/cgi-bin/ecotox\\_search](http://www.epa.gov/cgi-bin/ecotox_search)). This reflects a long-standing scientific consensus that effects on survival, growth, and reproduction clearly have the potential to affect the species population (Mount and Stephan, 1967).

In expressing its concerns about sublethal effects, NMFS cited two white papers prepared by its scientists (Johnson, 2000; Meador, 2000) that not only specifically examined the literature concerning this question, but more importantly concluded that there was sufficient information available to relate the specific amount of toxicant consumed by the juvenile salmonid in its food (i.e., the dose) to adverse sublethal effects. The availability of these analyses enabled the study team to address the issue of dietary toxicity to juvenile salmon.

## 2.2 Objectives

The purpose of this report is to address the issues raised by NMFS concerning the risks posed to juvenile salmon that consume estuarine prey contaminated with persistent and bioaccumulative chemicals<sup>6</sup>. Concern focused on certain compounds within three classes of chemicals that are well known for their environmental persistence, bioaccumulation, and aquatic toxicity at low concentrations: PAHs, PCBs, and DDT and its metabolites. These chemicals are generically referred to as contaminants throughout the remainder of this report. A second objective is to evaluate whether the dredging activities associated with the lower Columbia River navigation channel deepening project increase concentrations of these contaminants in the sediments, especially within the estuarine turbidity maximum (ETM) (Figure B-4). A third objective is to determine whether the proposed dredging activities increase access to these contaminants by endangered salmonids via dietary uptake (Figures B-5 and B-6) and if so, whether the amounts consumed while feeding are high enough to pose risks.

## 2.3 Scope

The issues posed by NMFS were addressed by conducting a preliminary risk assessment to evaluate potential risks based on the following information.

- Dietary toxicity of total PAHs ( $\Sigma$ PAHs), total PCBs ( $\Sigma$ PCBs), and DDT and its metabolites DDD and DDE ( $\Sigma$ DDT) to fish
- Toxicity of sediment-associated  $\Sigma$ PAHs,  $\Sigma$ PCBs, and  $\Sigma$ DDT to sediment-dwelling invertebrates.
- Sediment concentrations of contaminants in the lower Columbia River, as reported by state and federal agencies

A risk assessment was undertaken to address concerns about potential risks to salmonids and their prey that were posed in the NMFS Biological Opinion (1999). Contaminant risk is the probability of adverse

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<sup>6</sup> DDT (+DDD +DDE) and PCBs are termed PBTs (Persistent, Bioaccumulative and Toxic) Chemicals, and there is a national strategy for controlling sources of release of these chemicals (Davey 1999).

effects from a defined exposure or exposures (EPA, 1992<sup>7</sup>). A risk assessment would not have been possible without having data available that defined potential exposures and potential effects. Otherwise the risk assessment would have had to be qualitative rather than quantitative.

## **2.4 Report Organization**

The remainder of this report is organized into four sections that correspond directly to the risk assessment paradigm used nationally for more than 10 years (EPA, 1992). Section 3 includes the methodological details for the risk-based approach, including data sources, assumptions and uncertainties concerning the data, and data analysis methods. Findings are presented in Section 4. Uncertainties and assumptions underlying the findings are outlined in Section 5. Section 6 details the literature cited in this report.

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<sup>7</sup> EPA has formally defined ecological risk assessment as a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. (EPA, 1992).

### **3 RISK-BASED APPROACH**

#### **3.1 Introduction (Problem Formulation)**

##### **3.1.1 Issues**

Five issues were addressed in this analysis. Each is presented below as a question, along with its basis and rationale.

**Issue 1: Do risks increase in the lower Columbia River proceeding from below Bonneville Dam (RM >101) to the lower Columbia River estuary (RM 0-40)?**

If contaminant concentrations increase proceeding from below Bonneville Dam (RM >101) to the estuary (RM 0-40), then risks to estuarine-rearing salmonids, especially ocean-type chinook, may be greater when they rear for variable durations (days to weeks to months) in the lower river estuary (Aitkin, 1998).

**Issue 2: Are concentrations of organic carbon and fine particulate matter higher in the Columbia River ETM zone than in other locations, and could the potential suspension of fine particulates during channel dredging operations increase these risks?**

The ETM has been identified as being of special importance to the food web on which some life stages of salmonids depend, especially ocean-type juvenile chinook. The ETM is the site of a food web founded on microorganisms that consume the microscopic detritus produced upstream (e.g., dissolved and particulate organic carbon, including phytoplankton). Dissolved and particulate organic carbon (detritus) will precipitate upon mixing with seawater and reflux, by tidal action, in the lower river, until it is consumed or swept seaward. The ETM generates much of the secondary production in the Columbia River estuary (Simenstad, et al., 1990a,b, 1992, 1994a,b; Jay and Musiak, 1994). It occurs near the upstream head of saltwater intrusion, and within it water quality and biological productivity are highly interrelated. Microorganisms feed on the detrital flocs that form and settle during periods of weak currents. Various invertebrates (e.g., epibenthic zooplankton, and amphipods) feed on the aggregates by removing the microbial biofilms. In turn, they are fed upon by other invertebrates and certain life stages of juvenile salmon (see Figures B-4 and B-6). Theoretically, contaminant concentrations within the ETM should also be augmented because they tend to be sorbed to both dissolved and particulate organic carbon, reflecting their extreme hydrophobicity. Therefore, if the concentration of organic carbon increases in the ETM, the concentration of contaminants should also increase.

**Issue 3: Are chemical concentrations in the Columbia River navigation channel higher than those in nearshore sediments?**

In theory, navigation channel concentrations should be lower rather than higher because the fine particulates, which contain the highest concentrations of organic carbon and accordingly of contaminants (Karickhoff and Morris, 1985), are swept out of higher energy areas like channels and settle out in low energy, nearshore depositional areas.

**Issue 4: Are sediment concentrations high enough to pose risks to juvenile salmonids?**

This is the main question posed by NMFS. Its studies in the lower Duwamish River estuary (Seattle) and in the Hylebos Waterway (Port of Tacoma) suggest, based on multiple lines of evidence, that juvenile salmonids are not only being exposed to increased contaminants, but are responding immunologically and biochemically in ways that may signal stress and decreased fitness.

## Issue 5: Are sediment concentrations high enough to pose risks to the sediment-dwelling invertebrate prey of juvenile salmon?

If a significant fraction of the sediment-dwelling invertebrate prey of juvenile salmonids is affected at similar or lower contaminant concentrations than those directly affecting the juvenile salmon, then there could be indirect effects on the growth and hence the predation susceptibility of salmonid juveniles.

### 3.1.2 Description of the Overall Approach to Risk Analysis

This risk assessment relies on EPA's risk-assessment paradigm, as shown in Figure B-7, and a standard methodology for conducting aquatic ecological risk assessments (Parkhurst, et al., 1995). Figure B-7 shows that the risk assessment process begins with a problem formulation, which essentially defines the issues, how they will be addressed, and the assessment's scope. The scope of this project is to evaluate the potential risks to juvenile salmonids posed by contaminants that may be released during dredging in the navigation channel of the lower Columbia River, downstream of the confluence of the Willamette River and Columbia River at RM 101.

The substances assessed are mixtures of PCBs, DDT and metabolites, and PAHs. These chemicals are the sole focus of this risk assessment, based on previous studies. A variety of agencies have examined the effects posed by other water quality variables during and after dredging activities (e.g., Servizi, 1990) and these chemicals are the only ones that are currently considered to pose potential risk, mainly because of their bioaccumulative potential, persistence, and tendency to interfere with endocrine and immune systems (Dillon, et al., 1995; Varanasi, et al., 1993; Arkoosh et al., 2001). Each of the chemical groups of interest consists of many chemicals that have similar modes of action – so similar that it is possible to express their toxicity by summing their concentrations.

The compounds included in these summations are usually reported analytically as  $\Sigma$ PCB,  $\Sigma$ PAH, and  $\Sigma$ DDT (the sum of DDT, including its metabolites DDD and DDE and their isomers). The PAHs are often analyzed as individual compounds and then summed. For this report,  $\Sigma$ PAH was based on the 18 compounds used by Johnson (2000) in her paper on PAH effects (Table B-1) and reported analytically. Specifically, if only 8 of the 18 PAHs addressed by Johnson (2000) were reported analytically, only those 8 were included. It was assumed that the  $\Sigma$ PAHs,  $\Sigma$ DDT, and  $\Sigma$ PCBs reported analytically represented those compounds that occurred in the highest concentrations in each sediment sample.

**Table B-1. Low molecular weight (LMW) and high molecular weight (HMW) PAHs included in Johnson's (2000) analysis of the relationship between sediment PAH concentrations and the effects on >2-year old English sole (*Parophrys vetulus*).**

Analyte Number	Analyte Name	Molecular Weight Categorization
1	Biphenyl	LMW
2	Naphthalene	LMW
3	1-Methylnaphthalene	LMW
4	2-Methylnaphthalene	LMW
5	2,6-Dimethylnaphthalene	LMW
6	Acenaphthalene	LMW
7	Fluorene	LMW
8	Phenanthrene	LMW
9	1-Methylphenanthrene	LMW
10	Anthracene	LMW

11	Fluoranthene	HMW
12	Pyrene	HMW
13	Benzo(a)anthracene	HMW
14	Chrysene	HMW
15	Benzo(a)pyrene	HMW
16	Benzo(e)pyrene	HMW
17	Perylene	HMW
18	Dibenz(a,h)anthracene	HMW

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The next step in the risk assessment methodology is to characterize exposure by defining the frequency and magnitude of the concentrations that will be encountered by salmonid juveniles when feeding on epibenthic organisms in the lower Columbia River. In this report, exposure is characterized with a cumulative probability distribution, as shown in Figure B-8. It depicts the frequency and magnitude of contaminant concentrations measured in sediment samples from the three reaches studied: lower Columbia River estuary (RM 0-40), Columbia River to the Willamette River confluence (RM 41-101), and Columbia River from its confluence with the Willamette to below Bonneville Dam (RM >101). The probability distributions defining exposure convey considerable information about contamination. For example, as shown from the data in Figure 8, ΣPCB concentrations range from 0.05 parts per billion (ppb) dry weight (dw) to 28,000 ppb dw. Thirty-four percent of the measured concentrations are <12.5 ppb, 66% are >12.5 ppb, and the top 20% are above 150 ppb.

The objective of the third component of the risk-assessment paradigm, the effect characterization, is to characterize the effects that might be expected in juvenile salmon or other aquatic life upon exposure to different concentrations of contaminants. Data come mainly from laboratory tests of these substances, although in a few cases they are based on field data.

The effect characterizations for each contaminant were summarized using graphs like the one shown in Figure B-9. Figure B-9 presents four sets of criteria. The first consists of a set of responses occurring at different concentrations developed by Johnson (2000) for marine fish (English sole, *Parophrys vetulus*) exposed for at least 2 years to PAHs in sediments. The effects range from 54.1 ppb dw (preneoplastic foci of cellular degeneration in the liver) to 4,000 ppb dw (inhibited gonadal growth). In Johnson's (2000) analysis, note that as the sediment PAH concentration increases, the probability and severity of adverse responses also increase. A second criterion, proposed by Collier (2001), specifies 1,000 ppb dw as the approximate concentration at which no observed effects in marine fish have been noted. The third criterion (5,000 ppb dw), also proposed by Collier (2001), represents the lowest concentration where adverse effects have been observed in juvenile salmon. The fourth criterion is the regional screening guideline (15,100 ppb dw) used by regional, federal, and state agencies to evaluate whether sediments require further testing and assessment (Corps, et al., 2001). Effects of all the substances were characterized in the foregoing manner.

The final step in the risk assessment is the risk characterization. The basic approach was to graphically compare the probability of exposure to the probability of effect see Figure B-40. Risk potential was expressed in terms of the percentage of samples from (1) the Columbia River navigation channel and (2) all sediments that exceeded one or more of the screening criteria.

### 3.1.3 Conceptual Model

Three separate models were used to demonstrate how toxic substances in sediments could ultimately affect juvenile salmon adversely. The first model (Figure B-5) shows the expected fates of the contaminants in water and sediments and their uptake by juvenile salmonids. It mainly reveals that

contaminants will partition among water, sediment, and prey, and that juvenile salmonids and their prey will take up contaminants via both the water and dietary pathways. The contaminants are bound to the suspended particles and to the sediments. Figure B-6 shows the sources of suspended particles and other foods eaten by prey of juvenile salmonids. Figure B-10 depicts the Columbia River ecosystem-based conceptual model for juvenile salmonids. It identifies contaminants as one of the factors that have the potential to affect survival of juvenile salmonids via effects on their physiology.

## **3.2 Exposure Characterization**

### **3.2.1 Data**

#### **3.2.1.1 Justification for Reliance on Sediment Data Versus Tissue Residue or Water Concentration Data**

Data on chemical concentrations in sediment were used preferentially over data on tissue residues and water concentrations for several reasons. For at least the past 15 years, data concerning total contaminant concentrations in sediment have been collected preferentially for evaluating the risks of dredging and disposal (Corps, et al., 2000). This preference reflects professional convictions that sediments usually are less variable, easier to collect, and more ecologically relevant to sediment-dwelling organisms, which are preyed upon by a variety of fish and invertebrates, compared to contaminant concentrations in tissue and water. Perhaps most importantly, there is a direct relationship between sediment exposure and effects on sediment-dwelling organisms (Di Toro, et al., 1991), whereas more uncertain extrapolations (models) are required to estimate sediment risks based on water and tissue concentrations of PCBs, DDT, and PAHs. For these reasons, substantial amounts of sediment data have been collected over the years, and they were sufficient to accomplish this detailed risk assessment.

If sufficient data concerning contaminant concentrations in epibenthic invertebrate prey of juvenile salmon had been available, it would have been preferred, even more than sediment data, because it would have been most relevant to the risk issues. To our knowledge, however, data are available from less than a dozen samples from one site near Sand Island in the lower river estuary (NMFS, unpublished data), too few to support a comprehensive risk assessment. Most available tissue residue data for the lower Columbia River pertain to subadult/adult specimens of fish and invertebrates. These data were considered inappropriate for this specific risk assessment for several reasons: the exposure histories of the specimens were unknown; there were insufficient data to normalize for exposure duration or age; and lipid contents and diet were unknown.

Exposure history is important because a specimen caught in one location could have been exposed at a different site. In other words, its exposure may be unrepresentative of sediments at its capture site. Specimen age is also important because the longer the exposure duration, the greater the residues accumulated, since PCBs and DDE cannot be materially metabolized or depurated (Wisconsin Division of Health and Wisconsin Department of Natural Resources, 1997). Further, bioaccumulation magnitude depends greatly on lipid content (Lake, et al., 1990), which varies greatly in the many types and ages of specimens sampled. Finally, the diet of the specimens sampled needs to be specific to the pathway of interest, namely sediment  $\Rightarrow$  invertebrate prey  $\Rightarrow$  juvenile salmon (Figures B-5 and B-6). The diets of the sampled specimens cannot be assumed to reflect this pathway.

Surface water concentrations of the contaminants presumably contribute marginally to the tissue residues measured in the epibenthic invertebrates eaten by juvenile salmonids because their concentrations are extremely low and because surface water and sediment concentrations cannot be presumed to be in equilibrium. McCarthy and Gale (1999) estimated dissolved water concentrations in the picogram-per-

liter range. Such low surface water concentrations are believed to be less important in determining exposure than concentrations in the particulate and interstitial water phases of the sediments (Brueggeman, et al., 1984; Di Toro, et al., 1991) for these particular contaminants<sup>8</sup>. Moreover, sediment concentrations cannot be estimated readily from water concentrations without modeling other chemical and hydrological variables.

### **3.2.1.2 Data Sources**

The data used in the exposure characterization were collected from three basic sources: WSDOE, the Corps, and a federally funded study conducted by Tetra-Tech, Inc., for the Lower Columbia River Bi-State Committee. The data used for this risk assessment are available from Parametrix, Inc., 1600 S.W. Western Blvd., Suite 165, Corvallis, OR 97333. Version 3.0 of WSDOE's Sediment Quality Information System (SEDQUAL 3.0) (WSDOE, 2000) was the first database queried concerning sediment data for the lower Columbia River and Willamette River. It represents a compilation of data entered by a wide range of users, based on templates available on SEDQUAL's Web site (<http://www.ecy.wa.gov/programs/tcp/smu/sedqualfirst.htm>). Most of this database represents studies of localized hotspots, i.e., known point sources of contamination.

SEDQUAL 3.0 provided a large amount of applicable data, but not all data were relevant to this assessment. Version 4.0 of SEDQUAL (WSDOE, 2001) was released midway through this project, and its data were compared to Version 3.0 to determine whether it contained any new data. No additional data applicable to the exposure characterization were provided in Version 4.0, but it was used as a Quality Assurance/Quality Control (QA/QC) tool (see Section 3.2.1.3 below). Accordingly, data were sought from other sources.

More data were known to have been collected in the lower Columbia River than were included in SEDQUAL. To augment the database, Corps data were used. Corps data were available for a large number of sites along the Columbia River and its tributaries, but only data that were relevant to the issues addressed in this Biological Assessment were used. The data included in-channel sediment samples (Corps, 1999) and recent sediment samples collected in the estuary as part of maintenance dredging (Corps, 2001). The majority of Corps sediment data, collected primarily to determine appropriate disposal sites for routine maintenance dredge spoils, was not included because it was not useful for indexing potential exposure of juvenile salmonids in the areas of the navigation channel proposed for dredging. Also, some Corps data were unusable because latitudes and longitudes were unavailable to pinpoint locations.

The final data set included in the exposure characterization consisted of the survey done as part of the lower Columbia River Bi-State Program (Tetra Tech, 1993). Tetra Tech, Inc. conducted a reconnaissance survey of contaminant residues in water, sediment, and biota (fish and crayfish) from the lower Columbia River (mainstem and backwater areas) below Bonneville Dam. These data were downloaded from the lower Columbia River Estuary Program Web site (LCREP, 2001). To supplement data collected in the 1993 study, Tetra Tech, Inc. later surveyed contaminants in the backwater areas of the lower Columbia River (Tetra Tech, 1994). Data collected from the backwater study were already included in the SEDQUAL 3.0 data set and therefore did not need to be added to our database.

In summary, the data available in SEDQUAL identified sediment concentrations for localized nearshore hotspots. The Corps data addressed the main shipping channels, where the proposed dredging would

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<sup>8</sup> Excepting PAHs with log<sub>10</sub> octanol-water coefficients of 3-5, for which water exposure is the most important uptake pathway (Thomann 2001).

actually occur, and supplemented the estuarine data contained in SEDQUAL. The Bi-State Surveys (1993 and 1994) are unique because they appear to be unbiased, systematic samples of sediments throughout the lower Columbia River and do not focus on hotspots. Accordingly, they may index contamination generally throughout the lower Columbia River system.

### **3.2.1.3 Materials and Methods**

#### **SEDQUAL**

To retrieve information from SEDQUAL, both a station group and two chemical groups were constructed. The station group made use of the Geographic Information System interface provided in SEDQUAL to retrieve stations only from below RM 145 (Bonneville Dam) to the mouth of the Columbia River. It also included data for the lower Willamette River, specifically the reach from Portland Harbor to the confluence, including Columbia Slough<sup>9</sup>. The chemical group option was used to access all applicable DDT and PAH sediment concentrations. The other chemical of interest (PCBs) could be queried without use of this option.

Data from SEDQUAL were modified after export to Microsoft Excel® spreadsheets. If the unit of concentration was parts per million (ppm), it was converted to parts per billion by multiplying by 1,000. Additionally, if samples had microgram per liter (µg/L) as the unit of concentration, they were deleted from the data set. Values recorded as undetectable were halved to approximate the true concentration, which lies somewhere between the detection limit and zero (Gleit, 1985). If there were multiple, same-day samples for the same location, the sediment concentrations were averaged (arithmetic) and their qualifier codes combined (except for the unknown or null value qualifier, “#”). In a few instances, all limited to the Willamette River, values from certain jurisdictions were eliminated from the data set if they had multiple samples at the same spot collected on the same day, but varied dramatically in their sediment concentrations. Such values were deleted because the nature of the chemicals queried is such that they do not biodegrade quickly, and the data were considered suspect as a result. Concentrations were deleted only if they repeatedly varied by up to a factor of six for the same chemical on the same day at the same station (e.g., 930 vs. 6,100 ppb dw for 4,4'-DDT). It was only necessary to eliminate certain ΣDDT and ΣPAH values from the Willamette River. No data were eliminated from the Columbia River data sets.

Concentrations labeled in SEDQUAL as percent fines, total organic carbon (TOC) and total PCBs were used as output, but data for DDT and metabolites and PAHs were modified before being used. Fine particulate matter was queried as “Percent Fines,” TOC as “Total Organic Carbon,” and ΣPCBs as “Total Polychlorinated Biphenyls.” For exposure characterization, ΣDDT consisted of the sum of 2,4'- and 4,4'-congeners of DDE, DDD, and DDT<sup>10</sup>. When one of these was reported as undetected, its concentration was set equal to one-half its detection limit (Gleit, 1985), and this value was added to those of the detected ones. If a congener was not analyzed for, it was ignored in the summation. This procedure was conservative and probably tended to overestimate the concentrations of undetected analytes because most jurisdictions did not analyze for the 2,4'-DDE, DDD, or DDT congeners; only Tetra Tech, Inc. (1993), sampled for all six.

The remaining chemical group, which required preprocessing prior to risk analysis, consisted of the 18 individual PAHs considered in Johnson's (2000) characterization of PAH effects on marine fish (Table B-

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<sup>9</sup> Columbia Slough is an urban tributary in north Portland that enters the lower Willamette River near its confluence with the Columbia River. It was only included here to provide further comparison to chemical concentrations observed in the lower Willamette and lower Columbia Rivers.

<sup>10</sup> A congener here is a compound having the same basic structure as others in the same class of compounds.

1). Most study sponsors analyzed for 12 of the 18 compounds. Data were summed for all PAHs analyzed. When a compound was reported as undetected, its value was set equal to one-half the detection limit and added to the full value of the detected ones. If a PAH from Johnson's Table B-1 (2000) was not analyzed for, it was ignored in the summation. For both  $\Sigma$ DDT and  $\Sigma$ PCB values, the qualifier codes that existed for each part of the total were conserved and denoted for the summed value. If none of the analytes making up the summed value had qualifiers, the “#” symbol was denoted. If one or more of the analytes was qualified, the “#” symbol was dropped and only the qualifiers denoted.

### **U.S. Army Corps of Engineers**

Corps sediment data were obtained from several documents available on the Internet (Corps, 1999, 2001). Sediment data were entered into Microsoft Excel® spreadsheets using the SEDQUAL format. As described in the preceding subsection, undetected concentrations were halved to approximate the true concentration. Additionally, data collected in the navigation channel versus nearshore were identified based on maps provided with the data on the Internet or by Corps personnel. The in-channel data set consisted of the mouth of the Columbia River sediment evaluation data collected in September 2000 (Corps, 2001), as well as Appendix B of the EIS (Corps 1999) data collected in June 1997 (without out-of-channel stations 05, 06, 07, 57, 75, 75A) (Siipola, pers. Comm.).

The in-channel data from Appendix B of the EIS (Corps, 1999a) consisted of percent fines, TOC, the sum of PCBs, the sum of 4,4'-DDE, DDD, and DDT, and the sum of 12 PAHs. The  $\Sigma$ DDT and  $\Sigma$ PAH data were summed in the same manner as the SEDQUAL data. The qualifier codes that existed for each part of the total were conserved and combined into one qualifier code for the total value (except for the unknown or null value qualifier, “#”). If a congener was not analyzed for, it was ignored in the DDT summation. Also, if a PAH from Johnson's Table B-1 (2000) was not analyzed for, it was ignored in the PAH summation. The PCBs sampled by the Corps (1999a) were Aroclors 1016, 1221, 1232, 1242, 1248, 1254, and 1260. If all Aroclors were undetected, one-half of the detection limit was entered as the value to be plotted. In the one instance where PCBs were detected, the sum of the two detected Aroclors was adopted as the value.

The procedures described above were applied to the four remaining Corps data sets used: Columbia River mouth, Baker Bay/Illwaco Channel, Chinook Channel, and RM 29-34 (Corps 2001). These data were collected in support of maintenance dredging in the estuary. The same seven Aroclors (1016, 1221, 1232, 1242, 1248, 1254, and 1260) that had been analyzed for in Appendix B were analyzed for in these evaluations. No Aroclors were detected in any of the samples, so one-half of the detection limit was plotted as the  $\Sigma$ PCBs concentration.

### **Bi-State Surveys**

Sediment data collected by Tetra Tech, Inc., for the Bi-State Survey were downloaded from the Lower Columbia River Estuary Program Web site (LCREP, 2001). The resulting Microsoft Excel® spreadsheets contained multiple columns representing a wide range of studies. Only data from the Bi-State Surveys were used, and they were organized into the database's standard format. Undetected concentrations were treated as described above. Data used consisted of percent fines, TOC, the sum of 2,4'- and 4,4'-DDE, DDD, and DDT, and the sum of 12 PAHs. The  $\Sigma$ DDT and  $\Sigma$ PAH data were summed as described previously. The same seven PCBs that were sampled for the Corps (1999, 2001) sediment evaluations were sampled for this survey. If a PCB was undetected, one-half the detection limit was specified as its value. When one congener was detected in a sample, it became the  $\Sigma$ PCB value. If multiple congeners were detected, their concentrations were summed to compute  $\Sigma$ PCB. Data qualifiers for  $\Sigma$ DDT and  $\Sigma$ PCB were applied as described above.

Three measurements were adjusted because they were regarded as outliers, reflecting either matrix interference, inconsistency with all other measurements from the same and nearby sites, or both<sup>11</sup>. For station D1 (Baker Bay), one  $\Sigma$ DDT value was adjusted from 60 to 6.0 ppb and one  $\Sigma$ PCB value from 125 to 12.5 ppb because all other sampling in the area of Baker Bay showed nondetects and the Bi-State samples showed evidence of matrix interference<sup>12</sup>. Specifically, all Corps (2001) sampling in the Ilwaco-Baker Bay region (n=9) has failed to detect  $\Sigma$ PCBs (DL = 10-15 ppb dw), and the  $\Sigma$ DDT values have ranged from 0.9 – 2.6 ppb dw. For station E9, the value for 4,4'-DDT was adjusted from 100 to 1.0 ppb dw because all other  $\Sigma$ DDT congeners measured at this site were undetected (DL = 2 - 3 ppb dw).

### **3.2.1.4 Data Quality Assurance/Quality Control**

#### **SEDQUAL**

WSDOE does not conduct any additional quality control (QC) or quality assurance (QA) of the data that study sponsors load onto the SEDQUAL database. As a result, some of the data in the templates may have been entered incorrectly before being sent to WSDOE, but this is impossible to determine without verifying data authenticity with the study sponsor. For this reason, some values that were inconsistent with other data for the same location were eliminated as described above; otherwise the data were used as is.

WSDOE sometimes uses some of the available data sets in SEDQUAL to derive sediment quality standards. To determine which data can be used to derive standards, each study has a QA level code assigned to its data if its overall quality has been certified by WSDOE personnel (Personal Communication, Martin Payne, WSDOE, 2001). If the study's overall quality has not been evaluated, it receives a U (unknown) QA level code. Approximately one-third of the SEDQUAL data used for this report had been assigned the U code. The QA codes were not consulted in establishing the data used in these analyses.

After being exported from the SEDQUAL 3.0 database and modified in Microsoft Excel® spreadsheets, all data modifications were checked by Parametrix, Inc. As a further check, the exact same procedures were performed on the data set that was output from SEDQUAL 4.0 and compared with the data from SEDQUAL 3.0 to ensure that the modifications were performed correctly.

#### **Corps**

The Corps data were taken directly from Corps documents available on its Web site and entered into Microsoft Excel® spreadsheets by Parametrix, Inc. personnel. After the data were entered into spreadsheets and modified as described above, all data entry and modifications were checked by Parametrix personnel.

#### **Bi-State Surveys**

The sediment data collected by Tetra Tech, Inc., for the Bi-State survey were downloaded and saved as a Microsoft Excel® spreadsheet at the Lower Columbia River Estuary Program Web site, so the data did not have to be entered by Parametrix personnel. After the spreadsheets were organized in the same

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<sup>11</sup> Personal Communication from Dr. Steve Ellis to Mr. Mark Siipola, Corps.

<sup>12</sup> Matrix interference occurs when other substances in the sample interfere (in this case positively) with quantification of the substance. When positive, it has the effect of raising the detection limit.

format as the other data sets and modified as described above, all data entry and modifications were checked by Parametrix personnel.

### **3.2.1.5 Data Analysis**

To address the five issues, the distributions of the data in terms of the frequency and magnitude of sediment were processed as follows. After the QA/QC procedures were complete, all data for each chemical class or sediment property were placed into one comprehensive Microsoft Excel® spreadsheet. Data that had been collected in the channel (Corps 1998) and at the Columbia River's mouth (Corps, 2001) were coded to compare channel and nearshore values. Also, data that had been collected from the Columbia Slough and lower Willamette River were segregated from mainstem Columbia River data.

River miles were assigned to each station in the mainstem, and the data were broken out into three reaches of the Columbia River: RM 0-40, 41-101, and >101. River mileages for the Willamette River and the Columbia Slough were separately distinguished with codes. RM 0-40 has been designated in this report as the lower estuary. It represents a large open area from the mouth up to near Puget Island, as shown in Figure B-11. The middle reach extends from RM 41 to the Willamette River confluence (RM 101; Figure B-11 and B-14), and the upper reach extends to the uppermost end of tidewater, near Bonneville Dam (RM 145; Figure B-14). For each river reach, the concentrations measured in the channel and those measured at all sites, channel plus nearshore, were ranked in terms of percentage occurrence from lowest to highest concentration and expressed as cumulative frequency distributions (Figure B-8).

## **3.3 EFFECTS CHARACTERIZATION**

The goal of an effects characterization is to define the range of effects expected from each of the three chemical classes. Effects here are limited to adverse effects, and adverse effects do not necessarily include all changes in the organism in response to contaminant exposure. Responses may reflect changes in some attribute of the animal's biochemistry or physiology, which are not necessarily adverse effects. Adverse effects can be sublethal or lethal, but both types need to ultimately affect attributes believed to influence the viability of species populations – namely growth, survival, and reproductive success. An adverse, sublethal effect is one that leads to reduced growth, survival, or reproductive success. Another goal is to define each chemical's no adverse effect threshold, which constitutes the concentration below which there appears to be negligible risk. Each threshold is important because it is used to screen all exposures for risk potential. Risk occurs when an exposure exceeds a threshold. Exposures creating potential risks require further assessment, whereas those below the threshold pose insignificant (negligible) risks and usually do not receive further evaluation.

The published literature and the database Ecotox (EPA, 2001) were surveyed to identify what dietary concentrations of the contaminants adversely affected fish. Emphasis was placed on juvenile salmonids to the extent such data were available; however, data on effects on fish and invertebrates were used to evaluate risks to sediment-dwelling invertebrates.

The categories of chemical concentrations shown in Table B-21 constitute regional sediment screening guidelines for dredged material that were developed by state and federal agencies (Corps, et al., 2000). The guidelines are based on correlations between sediment concentrations and effects on a variety of organisms, and they are meant to protect fish, their prey, and all other aquatic life<sup>13</sup>. Each value represents a threshold that triggers further testing and evaluation when exceeded (Corps, et al., 2000).

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<sup>13</sup> Primarily amphipods, larvae of bivalve molluscs or echinoderms, and polychaetes.

When a concentration falls below this threshold, the sediment is suitable for unconfined, open-water disposal at designated sites, provided no other chemical exceeds its respective screening guideline.

There are other sediment standards available, but they apply to uses of the water that are different from dredging activities. For example, the bioaccumulation screening values listed in Table 5-1 of Corps, et al., (2000) are applied when risks to human health from consumption of contaminated fish is an issue. In terms of other sediment standards, WSDOE has developed a marine sediment standard for total PCBs of 12,000 ppb organic carbon. This would be equivalent to 240 ppb dw for a sediment containing 2% TOC. This standard represents the Sediment Cleanup Screening Level/Minimum Cleanup Level for marine hazardous waste sites (WSDOE, 2001).

The toxicological endpoint for analysis of PCB and DDT effects was the residue associated with effects in a toxicity test. Usually, this was the lowest observed effect residue (LOER) in the fish's tissues. The LOERs had to be converted to an equivalent sediment concentration. To do this, it was assumed that each contaminant's residue in the fish's tissues was in equilibrium to that in the prey and then to the corresponding sediment concentration (Figure B-5; Di Toro, et al., 1991). This is a conventional assumption for such substances (EPA, 2000), but there are uncertainties, as discussed in Section 5. To express LOERs in terms of dry weight sediment concentrations, the following equation was used (following Meador 2000):

$$\text{Sediment Concentration (ng/g dw)} = (\mu\text{g/g lipid} \times \text{TOC} \times 1000) / \text{BSAF}$$

The TOC was assumed to be the arithmetic mean TOC content for the reach in question (i.e., RM 0-40, 41-101, >101). This conversion was not performed for the PAH effects data because Johnson (2000) expressed effects in terms of sediment dry weight.

### **3.3.1 PCBs**

An analysis by Meador (2000a) was used to represent potential dietary effects of PCBs because it appeared to be the most recent and complete analysis of potential relationships between PCB tissue residues and effects on juvenile salmonids. The data used by Meador (2000) are graphically depicted using the concentration-response relationship shown in Figure B-12. Based on the TOC normalization, PCB effects range from 10.6 to 17,363  $\mu\text{g/g}$  (ppm) TOC. Meador (2000) suggests the effect threshold should be set equal to effects associated with 10% or more of the endpoints, i.e., approximately 15.0  $\mu\text{g/g}$  TOC (Figure B-12).

### **3.3.2 DDT and Metabolites**

Data are limited concerning either sublethal or chronic effects of DDT and metabolites on the salmonid life stages found in the Columbia estuary, based on a screening of toxicity values from EPA's Ecotox database: [http://www.epa.gov/cgi-bin/ecotox\\_search](http://www.epa.gov/cgi-bin/ecotox_search). Most of the available data concern toxicity from water-only exposures, and there are few data defining toxicity from dietary exposures or the tissue residues found in the salmon. However, there is a regional sediment screening guideline of 6.9 ppb dw (Table B-2), based on sediment toxicity tests of several invertebrates.

The best study appears to be a 612-day chronic exposure of cutthroat trout (*Oncorhynchus clarki*) to DDT by the U.S. Fish and Wildlife Service (Allison, et al., 1963; 1964). It is considered most relevant to the exposure scenarios typical of the Columbia estuary where juveniles, mainly smolts or older, are the life stages being exposed. Equally important, the exposures lasted for months and toxicity was expressed in terms of the residues measured in the specimen's tissues. The test was started with 21-month-old specimens and encompassed reproduction. A variety of endpoints were measured, such as growth,

mortality, reproductive success, histopathology, and resistance to stress, including disease and temperature stress. Fish were either fed various concentrations of DDT in the diet once a week or were placed once a month for 30 minutes in water containing various concentrations of DDT. These simulated intermittent, but long-term water and dietary exposures. Tissue residues of total chlorinated hydrocarbons were measured frequently throughout the experiment. It was possible to relate these residues to mortality, which combined with growth effects, was the most sensitive endpoint. It was assumed that measurement of total chlorinated hydrocarbons was equivalent to measuring  $\Sigma$ DDT and its metabolites because fish metabolize DDT to DDE fairly rapidly, but DDE resists metabolism and persists within the organism. Allison, et al., (1963, 1964) reported that mortality of fish given DDT baths became statistically different from controls when tissue residues reached between 1,900 and 4,200 ppb wet weight (ww)<sup>14</sup>. Likewise, mortality became significantly different from controls (i.e., tissue residues between 5,600 and 7,000 ppb ww) when fish were fed DDT in their diet. These are regarded as estimates of LOERs. The difference between water and dietary LOERs may reflect different assimilation efficiencies, as discussed by Gobas, et al., (1989).

Other studies of DDT's effects on juvenile salmon were consulted, including Halter and Johnson (1974) and Dill and Saunders (1974). Unfortunately, it is difficult to extrapolate these studies to exposures characteristic of the Columbia estuary, principally because they tested life stages that either do not occur or do not rear in the estuary, specifically the shipping channel (embryos, alevins, fry). The life stages and species of interest in this risk assessment stop and rear in the Columbia estuary prior to migrating seaward. The life stages comprise pre-smolt, smolt and post-smolts of chinook, coho, cutthroat, and steelhead.

**Table B-2. Regional Sediment Screening Guidelines for Judging Suitability of Dredged Material for Unconfined Disposal**

Chemical	Guideline, µg/kg dw
$\Sigma$ PCB	130
$\Sigma$ DDT	6.9
$\Sigma$ PAH	15,100 <sup>a</sup>

<sup>a</sup>Based on the screening levels shown in Table 5-1 of Corps, et al., (2000) that matched the PAHs identified in Table 1 of Johnson (2000).

Source: Table 5-1 of Corps, et al., (2000).

The LOERs were converted to equivalent sediment concentrations by assuming that sediments were in equilibrium with the tissues of the juvenile salmonids living in the lower Columbia River. This is a worst-case estimate of exposure because equilibrium is unlikely, especially close to the mouth, where water from the Pacific Ocean dilutes estuarine water and the degree of dilution fluctuates with tides and river flow. The field-derived biota-sediment accumulation factor (BSAF) developed by Wong, et al., (2001) of the U.S. Geological Survey, was used. The BSAF is defined as follows:

$$\text{BSAF} = C_{t(l)} / C_{s(\text{foc})}$$

<sup>14</sup> The values are taken from Figure 1 and Table 9 of Allison et al. (1964) and the paper's discussion of which concentrations were significantly different from controls and when. Allison et al. (1963, 1964) did not indicate whether the units were wet or dry weight. Wet weight was assumed because most papers written in that era used wet weight unless another unit was specified. If units were actually dry weight, then tissue residues reported by Allison would be approximately 20% of those reported, i.e., between 380 and 840 ppb ww in tissue or 1120 to 1400 ppb ww in food.

where  $Ct(i)$  refers to the tissue concentration normalized to the lipid content of the tissue and  $Cs_{(foc)}$  refers to the sediment concentration of DDE, normalized to the sediment's total organic carbon content.

Based on field-studies of many species of fish across the United States, Wong, et al. (2001) found that the BSAF for DDE (median = 8.6) was the highest reported for the substances examined. Using the LOER range of 1,900 to 5,600 ppb ww and this BSAF, the sediment concentrations corresponding to the LOERs were estimated to be as low as 46,027 ppb TOC to as high as 135,659 ppb TOC. These sediment concentrations were adjusted further for the median TOC observed within each river reach. For example, within RM 0-40, channel and channel plus nearshore sediments averaged 0.05% and 0.76% TOC, respectively. Upstream sediments were virtually identical in TOC to those downstream. For example, channel sediments averaged 0.24% and 0.06% within RM 41-101 and RM > 101, respectively, and nearshore and channel sediments averaged 0.52% and 0.58%, respectively. Moisture content was assumed to be 20% and the lipid content of the juvenile salmon was assumed to be 2.4%, the median value in data summarized by Meador (2000a).

### 3.3.3 PAHs

Three screening criteria and one study of concentration-response relationships involving PAHs and fish were used to define the range of potential effects to juvenile salmon. The Regional Screening Guidelines (Corps et al., 2000) specify two screening levels for PAHs – 5,200 ppb ( $\mu\text{g/g}$ ) dw for low molecular weight PAHs and 12,000 ppb dw for high molecular weight PAHs. These equated to 15,100 ppb dw as  $\Sigma\text{PAH}$  when only compounds analyzed by Johnson (2000) were totaled. Johnson's (2000) analysis of 18 low and high molecular weight PAHs (Table B-1) was used because it represented NMFS' most recent analysis of potential relationships between sediment concentrations and effects on fish. It is specific and limited to English sole and similar sediment-dwelling species that have received long-term exposure (more than 2 years) to PAHs. Consequently, it cannot be applied directly to juvenile salmon, which spend an average of 25 days or less (Aitkin, 1998) feeding on a combination of epibenthic and pelagic prey as they grow in the estuary. However, it can be used as a reference point. Johnson (2000) based her analysis on a variety of changes that NMFS observed in laboratory and field populations of sole, which they assumed were caused only by PAHs. These responses ranged from preneoplastic foci of cell degeneration in the liver, correlated with a sediment PAH concentration of 54 ppb dw (2% TOC assumed), to inhibited gonadal growth at a sediment PAH concentration of 4,000 ppb dw (Table 2-4; Johnson, 2000). The same data are plotted in Figure B-13 in the manner that they are used in risk assessment.

Dr. Tracy Collier (2001) has proposed two screening criteria for  $\Sigma\text{PAH}$  effects on fish. Based on NMFS analysis of the existing data, Collier suggested that a  $\Sigma\text{PAH}$  sediment concentration of 5,000 ppb dw represents the lowest observed effect concentration (LOEC) for juvenile salmon exposed to PAHs in sediments, and 1,000 ppb dw approximates the no observed effect concentration (NOEC) for marine fish like English sole. All of the foregoing sediment concentrations were used to assess the potential for and magnitude of risks to juvenile salmon.

## 4 FINDINGS (RISK CHARACTERIZATION)

The risk characterization is presented in terms of the five issues (questions) first discussed in Section 3 (Risk-Based Approach) of this report:

- **Issue 1:** Do chemical exposures increase in the Lower Columbia River from Bonneville Dam (RM >101) to the Lower River estuary (RM 0-40)?
- **Issue 2:** Are concentrations of organic carbon and fine particulate matter higher in the estuarine turbidity maximum zone relative to other locations, such that suspension of fine particulates during channel dredging operation could augment risks?
- **Issue 3:** Are chemical concentrations in the navigation channel higher than those in nearshore sediments?
- **Issue 4:** Are sediment concentrations high enough to pose risks to juvenile salmon?
- **Issue 5:** Are sediment concentrations high enough to pose risks to the invertebrate prey of juvenile salmon?

### 4.1 Issue 1

**Finding: Contaminant concentrations decrease rather than increase proceeding from Bonneville Dam to the mouth and are lowest in the estuary. The basis is described in the remainder of this subsection.**

The first issue addressed here is whether chemical exposures increase on the Columbia River from above Bonneville Dam (RM >101) to the lower estuary (RM 0-40). Because juvenile salmon tend to rear most extensively in the lower river and estuary, it is important to learn whether they are receiving greater chemical exposure in the lower river. Increased exposure increases the potential for risk but is not necessarily associated with risk. This issue was addressed by plotting sediment concentrations on the Columbia River from the upper reaches of the project study area to the mouth for  $\Sigma$ PCBs,  $\Sigma$ DDT, and  $\Sigma$ PAHs.

All chemical concentrations reported in this section have been normalized to dry weight rather than TOC in order to use all the sediment data that were available from SEDQUAL, Tetra Tech, Inc., and the Corps. In particular, reliance upon dry weights allowed use of all data for channel sediments whose TOC contents usually were too low – i.e., below the detection limit (usually 0.05%) – for normalization to TOC.

#### 4.1.1 PCBs

Sediment concentrations of  $\Sigma$ PCBs were all less than 130 ppb dw at all sites throughout most of the Columbia and Willamette River except for a few locations, as identified in Figures B-11 and B-14. Data presented in Section 4.2 indicate that elevated concentrations occur in nearshore depositional environments near to point or non-point sources of contamination. There was no evidence of concentrations increasing in the ETM (i.e., lower Columbia River estuary) (Figure B-11).

### 4.1.2 DDT and Metabolites

Concentrations of DDT and its metabolites (DDD and DDE) were lower in the estuary (<6.9 ppb dw) than farther up the Columbia River, and they were highest (up to and exceeding 60 ppb dw) in the lower Willamette River and the Columbia Slough, which drains an agricultural area near Gresham and urban, northwest Portland along its length (Figures B-15 and B-16). Therefore, there is no evidence that ΣDDT concentrations were higher in the ETM compared with upstream locations.

### 4.1.3 PAHs

Total PAH (ΣPAH) concentrations observed in the estuary appeared to be comparable to those found in the mainstem Columbia to above the Willamette River confluence and do not indicate any obvious augmentation in the region of the ETM (Figures B-17 and B-18). Most were less than 1,000 ppb dw and all were less than 5,000 ppb dw, which in turn are both less than the regional sediment screening guideline of 15,100 ppb dw (Table B-2). The ΣPAH concentrations recorded for the lower Willamette and its tributary Columbia Slough were considerably higher than those in the mainstem Columbia (Figure B-18).

## 4.2 Issue 2

**Finding: Organic carbon and fine particulate contents are not elevated in navigation channel sediments in the lower estuary, within which the ETM occurs. The basis for this conclusion is provided in the remainder of this subsection.**

Because organic carbon tends to be trapped temporarily within the ETM, any chemicals bound to these particles will likely behave similarly. The ETM zone may extend approximately from the mouth of the Columbia River to RM 10-16. Trapping occurs when charged particles bind to the ions in saltwater, precipitate, and reflux with the tides (Reed and Donovan, 1994).

The ETM within the Columbia estuary has been studied extensively. It is confined to the Columbia River's channel (Simenstad, et al., 1994), and mixes vertically (Baross, et al., 1994; Simenstad, et al., 1994a). Enhanced turbidity associated with the microbial decomposition of organic matter also is found there (Reed and Donovan, 1994). Owing to this turbidity, the ETM's productivity has been questioned compared to less turbid environments (Small and Morgan, 1994). However, the detrital floc represents an important food source for the invertebrates that live within (infauna) and upon (epibenthos) the bottom sediments. Zooplankton have been documented feeding within the ETM, due to the organic detritus there (Baross, et al., 1994; Reed and Donovan, 1994). *Corophium* amphipods are thought to be associated with the ETM, because they feed on the organic matter. Theoretically, these processes may increase exposure and therefore potential risk to salmon when the juveniles feed upon the invertebrates living with the ETM.

The question of whether concentrations of organic carbon and fine particulate matter increase in the ETM relative to other estuarine locations was specifically addressed. Both of these substances would tend to be suspended and could remain on the sediment's surface after channel dredging.

Maps depicting spatial changes in organic carbon suggest TOC is slightly augmented in the lower river estuary compared with most sediments upstream to Bonneville Dam (Figures B-19 and B-20). However, all sediments within the Columbia River study area are lower than those in the lower Willamette River (Figure B-20). Figure B-21 suggests the increased TOC is associated with shallow, nearshore depositional habitats rather than with channel sediments. This reflects the higher energy typical of the channel environment; the TOC particles usually are too small and light to settle in the high-energy environment of the Columbia channel. Instead, they settle in low-energy areas, which generally are close

to shore and in embayments. A comparison of TOC concentrations in nearshore versus channel sediments indicates that the latter are so sandy that 93% of them contain negligible TOC ( $< 0.08\%$ ) (Figure B-21). All sediments sampled in the lower Columbia River contain low concentrations of TOC: approximately 81% of the samples contained  $\leq 1\%$  TOC.

Organic carbon and fine particulate concentrations are known to be correlated; however, in the data analyzed, the correlation between fine particulate and TOC concentrations was only moderate ( $r=0.77$ ). The channel sediments are so sandy, compared with nearshore sediments, that 93% contained 4% or less fines (Figure B-22). Although nearshore sediments contained higher concentrations than the channel, fine particulate concentrations were still low; half the samples contained  $\leq 17\%$  fines (Figure B-22). The spatial distribution of fines within the lower Columbia shows that concentrations are very low in the navigation channel and higher at sites nearshore (Figures B-23 and B-24). Near the Willamette confluence, channel sediments still contain mostly low percentages of fines, but the lower Willamette and Columbia Slough contain much higher concentrations of fines (Figure B-24).

### **4.3 Issue 3**

**Finding: In all instances, concentrations of PCBs, DDT and metabolites, and PAHs in the channel were markedly lower compared with samples collected nearshore, as discussed in the following three subsections. These differences were consistent throughout the reaches studied, from the mouth of the Columbia River to near Bonneville Dam.**

#### **4.3.1 PCBs**

$\Sigma$ PCB concentrations increased from the mouth to Bonneville Dam, were consistently at the detection limit in channel sediments, and were lower below the Willamette River confluence than above it. In the lower estuary (RM 0-40),  $\Sigma$ PCB concentrations were at the detection limit (5-6.5 ppb dw) in the channel and only 12.5 ppb dw or less in nearshore sediments (Figure B-25). Upstream, all channel sediments were at the detection limit (5 ppb dw) except one. Most (78%) nearshore sediments also contained very low ( $\leq 12.5$  ppb dw) PCB concentrations, but 22% contained higher concentrations, up to 110 ppb dw (Figure B-26). Above the Willamette confluence,  $\Sigma$ PCB concentrations in the channel remained at the detection limit (5 ppb dw), but 34% of the sediments, all nearshore, ranged from 12.5 ppb dw up to 28,000 ppb dw. Ten percent of the sediments, all nearshore, exceeded 1,500 ppb dw (Figure B-27).

#### **4.3.2 DDT and Metabolites**

Concentrations of DDT and metabolites in sediments were always lowest in channel sediments and higher in nearshore sediments. Concentrations in nearshore sediments were lowest in the lower river estuary (RM 0-40) and distributed similarly from RM 40 to above the Willamette River confluence. In the lower river estuary (RM 0-40), 71% of the  $\Sigma$ DDT concentrations in channel sediments were at the detection limit of 0.28 ppb dw, and the remainder were at a higher detection limit of 3.0 ppb dw (Figure B-28). Nearshore sediments were higher, up to 6.5 ppb dw. Upstream (RM 41-101), all channel sediments were  $\leq 5$  ppb dw, compared to 70% of all (nearshore + channel) sediments being 6.5 ppb dw or less. Thirty percent of the samples, all collected nearshore, ranged up to 33 ppb dw (Figure B-29). Above the Willamette confluence (RM  $>101$ ), channel sediments remained at  $\leq 3.0$  ppb dw, whereas sediments nearshore ranged up to 30 ppb dw, the same range observed downstream (Figure B-30).

### 4.3.3 PAHs

Unlike  $\Sigma$ PCBs and  $\Sigma$ DDT, concentrations of PAHs in navigation channel sediments usually were above detection limits, though still significantly below concentrations observed in nearshore sediments. Nearshore sediments appeared to possess similar  $\Sigma$ PAH residues from the Columbia River mouth to above the Willamette confluence, although about 10% of RM 41-101 sediments were higher than those sampled upstream and downstream. In the lower river estuary (RM 0-40), concentrations ranged from 6.6-41 ppb dw in the channel (Figure B-31). Sixty-five percent of the nearshore samples exceeded the highest concentration observed in the channel, and nearshore sediments ranged up to 1,008 ppb dw, with one sample reporting 4,259 ppb dw. Between RM 41 and the Willamette confluence, 80% of channel sediments contained less than 41 ppb dw, and the maximum observed in the channel was 396 ppb dw (Figure B-32). Fifty percent of all sediment samples, all of them sampled nearshore, exceeded the maximum concentration observed in the channel. The two highest concentrations observed in nearshore sediments were 3,105 and 14,254 ppb dw (Figure B-32), three times higher than concentrations observed in the lower estuary (Figure B-31). Few channel sediments ( $n=5$ ) have been sampled above the Willamette confluence, and the levels are comparable to those observed at the majority of sites sampled all the way to the mouth (Figure B-33). Nearshore sediments contained almost as much PAHs as observed below the Willamette confluence; the maximum concentration observed nearshore at RM >101 ranked in the 90th percentile of all sediments sampled from RM 40-101 (compare Figure B-32 to Figure B-33).

## 4.4 Issue 4

**Finding: Contaminant concentrations in navigation channel sediments posed only negligible risks to juvenile salmon, whereas some nearshore sediments close to point sources of contamination posed risks. Risks associated with nearshore sediments were lowest in the lower estuary compared to upstream reaches (RM 40-145), as explained below.**

In this subsection, the sediment concentrations identified in Section 4.2 are compared with data that relate predicted effects to specific sediment concentrations. The comparisons enable judgments to be made concerning whether potential risks exist and if so, their magnitude. Because of the many assumptions and other uncertainties in the data on which they are based, the risks should only be regarded as potential. Whereas negligible risks typically warrant no further investigation, potential risks warrant further investigation. Such investigations usually focus on reducing uncertainties in the data and evaluating whether effects, suggested by laboratory tests of other species, are actually manifesting in the species in the field. In the United States, Europe, and Australia, risks affecting fewer than 5% of the species are regarded as negligible provided they do not affect species that are endangered, threatened, economically important, or ecologically keystone (Australian and New Zealand Environment and Conservation Council, 2000; Crommentuijn, et al., 2000; Stephan, et al., 1985). Keystone species are ones that will change a community's structure and function if removed.

Because concern here focuses on whether individuals within populations of endangered species are at risk, the question focuses on the level of impact required to place the population at risk. Meador (2000a) has suggested that effects on fewer than 10% of the individuals in a population of endangered or threatened species can be considered negligible, with exceedances requiring further investigation. In bioassays of substances, effects on 10% or fewer of the test specimens are considered comparable to controls (e.g., ASTM, 1998). Endpoints like the EC25 or IC25, which refer to the concentrations affecting 25% of the specimens, are widely used for estimating the NOEC level on the population (ASTM, 1998). Moore and Caux (1997) note that most (77%) NOECs from aquatic toxicity testing have been associated with effects on between 10 and 30% of the individuals in the test population. Conversely, most (62%) LOECs were associated with effects greater than or equal to 30% of the individuals. Based on these data,

there may be a consensus that effects on fewer than 10% of the individuals and perhaps as high as 25% may be indistinguishable from the effects normally observed in controls in toxicity tests. A higher level of mortality may be tolerated by salmon in nature, but it depends entirely on the cumulative magnitude of all sources of mortality. For example, in natural populations of some fish species, as many as 50 to 70% of the prospective spawners can be lost from the population (i.e., die), as a result of natural mortality, fishing, predation, toxicants, and other factors, without affecting spawning success and recruitment of the next generation. This is based on a study performed by Waller, et al. (1971). It is recognized, however, that agency decisions concerning appropriate levels of protection for species populations usually consider a complex set of social and political judgments and priorities in addition to scientific data.

The following risk characterization pertains only to Columbia River sediments proposed for navigation channel dredging. Nearshore sediments are expected to be unaffected by dredging activities associated with this project. If they do receive suspended solids as a result of channel deepening dredging, the overall level of contamination should remain unchanged because channel concentrations appear to be lower than nearshore concentrations. Dredging resuspends only minor ( $\leq 3$  milligrams per liter) amounts of particulate matter, and the suspended sediments contain a lower level of contamination than nearshore sediments.

#### **4.4.1 PCBs**

PCBs in channel sediments appear to pose negligible risk to juvenile salmonids from the mouth to above the Willamette confluence, based on comparison to all screening criteria concerning potential effects (Figures B-34 to B-36). Fewer than 10% of all sediments, all nearshore, posed potential risks to juvenile salmonids from the Columbia River mouth to the Willamette confluence, based on a comparison to the 10% screening criterion proposed by Meador (2000a) (Figures B-34 and B-35). Potential risks were higher above the Willamette confluence; about 20 % of the sediments – all nearshore – exceeded Meador's lowest screening criterion and 20% exceeded the regional screening guideline (Figure B-36). Based on these findings, dredging associated with this project is not expected to augment PCB risks in the navigation channel sediments because they are uniformly negligible in the lower river and estuary. Risks are higher in certain upstream, nearshore sediments. All these risk estimates are based on a number of important assumptions, which are discussed in Section 5.

#### **4.4.2 DDT and Metabolites**

Sediment concentrations calculated from the lowest observed effect residues discussed in Section 3.3.2 suggested that DDT and its metabolites would pose only negligible risks to juvenile salmonids from the mouth of the Columbia River to near Bonneville Dam (Figures B-37 through B-39). Moreover, there appeared to be a large margin of safety between the concentrations observed in the sediments and those associated with mortality in cutthroat trout, which appears to be the most sensitive salmonid tested to date with DDT.

The Regional Screening Guideline for DDT was lower than the LOERs calculated from Allison's (1963, 1964) studies (Figure B-37). The guideline also indicates that DDT in channel sediments should pose negligible risks, to aquatic life, but it suggests that about 25% of the sediments upstream of RM 41 –all nearshore –may pose risks.

#### **4.4.3 PAHs**

Risks from dietary ingestion of PAHs by juvenile salmonids were examined according to four criteria. The NMFS white paper (Johnson, 2000) analyzed laboratory and field data for English sole (Figure B-

40). She proposed a relationship between sediment concentrations of PAHs from 54 to 4000 ppb dw, and a variety of responses in English sole. Collier (2001) evaluated those data and suggested that 1,000 ppb dw was the NOEC for resident marine fish in the Columbia estuary. He also identified 5,000 ppb dw as the approximate LOEC for juvenile salmon based on evidence of DNA damage and immunosuppression. Assuming 5,000 ppb dw is the LOEC for juvenile salmonids, then by definition this would be the lowest concentration at which any effects should be observed, with higher concentrations being required to directly elicit more definitive effects, such as mortality. Sediment concentrations of  $\Sigma$ PAHs were evaluated in terms of all four criteria, plus the Regional Screening Guideline of 15, 100 ppb dw, recognizing that these criteria are based on laboratory and field studies elsewhere and may not be wholly applicable to conditions in the lower Columbia estuary.

In the lower estuary (RM 0-40), no navigation channel sediments exceeded any of the four criteria (Figure B-40). Sixty-four percent of all sediments, all nearshore, exceeded the lowest effect endpoint specified by Johnson (2000), but only 4% and 2%, respectively, exceeded Collier's (2001) proposed screening criteria for marine fish and juvenile salmonids. No sediment exceeded the Regional Screening Guideline. Upstream in the Columbia River (RM 41-101), all navigation channel sediments were below three criteria, and 10 of 11 samples were below all of Johnson's (2000) criteria (Figure B-41). Between RM 41 and 101, 80% of all sediments (all but one nearshore) exceeded Johnson's most conservative criterion; 12% exceeded Collier's (2001) marine fish criterion; and 2% exceeded Collier's (2001) juvenile salmon criterion. None exceeded the Regional Screening Guideline (Figure B-42). Above the Willamette River confluence (RM >101), all channel sediments were below all four screening criteria. Eighty five percent of all sediments, all nearshore, exceeded Johnson's (2000) minimum criterion, 6% exceeded Collier's (2001) marine fish criterion, and none exceeded either Collier's (2001) juvenile salmon criterion or the Regional Screening Guideline.

In conclusion, the preceding risk analyses suggest negligible risks to juvenile salmon resulting from exposure to the contaminants sorbed to sediments in the navigation channel that may be dredged for the proposed project. Only one channel sediment sample exceeded any of the screening criteria concerning potential effects. Nearshore sediments presented greater potential risks, but their magnitude depended greatly on the screening criterion. The most conservative criteria, set forth for PAHs by Johnson (2000), suggested the greatest risk potential, whereas the two PAH screening criteria set forth by Collier (2001) suggested negligible to low risks. Compared to the Regional Screening Guidelines, most sediments would be classified as having negligible risk potential. Based on a weight of evidence involving all four criteria, nearshore sediments in some locations pose potentially significant risks. Given the conservative assumptions in the risk calculations (see Section 5), which should tend to overestimate both exposure and potential effects, dredging activities on the navigation channel sediments are not likely to pose any significant risk potential to juvenile salmon rearing in the lower Columbia River.

## **4.5 Issue 5**

**Finding: Contaminant concentrations in navigation channel sediments posed negligible risks to salmonid prey, as explained below.**

The risks posed by the three contaminant classes to sediment-dwelling prey (benthos and epibenthos) of juvenile salmon were examined. A risk-based approach was used that defined exposure, effects, and risk potential.

There appears to be a consensus in the scientific community that equilibrium partitioning theory can be used to estimate exposures of sediment-dwelling invertebrates to contaminants bound within the sediments (Di Toro, et al., 1991; US EPA, 1994). This method was used because it accounts for the bioavailability of these substances in sediments and provides a cause-and-effect foundation for comparing

exposures to the existing data on aquatic toxicity. Based on this theory, contaminant concentrations in the sediments are assumed to be in equilibrium between the three sediment phases—water, solids, and tissues. Thus, one can assume that concentrations estimated for one phase are proportional to exposures in the other phases (Di Toro, et al., 1991). Therefore, it is possible to base exposures on either water concentrations, tissue concentrations, or sediment concentrations, whichever is available.

Exposures were estimated for  $\Sigma$ DDT using equilibrium partitioning. Specifically, a sediment-water partition coefficient was estimated using a  $\log_{10}$  octanol-water partition coefficient of 6.86 for DDT (Table I in Chiou, 1985) and the following equation from Di Toro, et al., (1991):

$$\log 10 K_{oc} = 0.00028 + 0.983 \times \log 10 K_{ow}$$

Then, the organic carbon-normalized sediment concentration (e.g., ng  $\Sigma$ DDT/g TOC) was divided by the antilog of the  $K_{oc}$  to yield the equivalent porewater concentration.

To gauge the reliability, the estimated porewater concentrations of  $\Sigma$ DDT were compared to the average concentration estimated by McCarthy and Gale (1999) for surface waters downstream of Bonneville Dam using semipermeable membrane devices.

Exposures for  $\Sigma$ PAH and  $\Sigma$ PCB were based on concentrations in the solid rather than aqueous phase because they could be compared to Swartz's (1999) effects threshold for  $\Sigma$ PAH and the study of McDonald et al. (2000) concerning  $\Sigma$ PCB. Both studies defined effect thresholds based on empirical testing and equilibrium partitioning studies by a variety of investigators.

Effects of the three contaminants were based on the scientific literature. For  $\Sigma$ DDT, effects data from US EPA's (1980) DDT water quality criteria document were used. Because the US EPA (1980) chronic criterion of 0.001  $\mu$ g/L is based on protection of brown pelicans from food chain exposure to DDT, a surrogate criterion for protecting fish and invertebrates had to be derived from their data using EPA guidelines (Stephan, et al., 1985). Accordingly, the final acute value of 1.1  $\mu$ g/L was divided by 2 to estimate the acute criterion (0.550  $\mu$ g/L), and this was divided by a generic acute-chronic ratio of 100 to estimate the concentration (0.0055  $\mu$ g/L) that is expected to protect 95% of the aquatic species. For  $\Sigma$ PAH, Swartz's (1999) threshold effect concentration of 290,000 ng/g TOC was used. This threshold equated to dry weight concentrations of 145 to 696 ng/g in channel sediments and 1508 to 2204 ng/g for both nearshore and channel sediments. The ranges reflect the different average TOC concentrations characteristic of each reach. The study by McDonald et al. (2000) was used because it provided a consensus threshold effect concentration for  $\Sigma$ PCB of 48 ng/g dw that reflected both empirical studies and equilibrium partitioning.

Risk potential to sediment-dwelling invertebrates was indexed by comparing the range of predicted exposures in Columbia River sediments, both those in the channel and nearshore, to the threshold effects concentrations (Figures B-43 through B-49). All concentrations of  $\Sigma$ DDT in both channel and nearshore sediment porewaters were below the concentration expected to protect 95% of aquatic species (0.0055  $\mu$ g/L) (Figures B-43 and B-44). The estimates of porewater concentrations appeared to be very close to dissolved concentrations estimated by the U.S. Geological Survey for surface waters downstream of Bonneville Dam, using semipermeable membrane devices (McCarthy and Gale, 1999) (Figures B-43 and B-44). All  $\Sigma$ PCB concentrations in channel sediments except one were below the threshold effect concentration developed by McDonald, et al. (2000) for the protection of estuarine and saltwater organisms (Figure B-45). About 10-25% of all sediments, all collected nearshore and above RM 40, exceeded the threshold effect concentration for  $\Sigma$ PCB, and about 15% of these samples exceeded the median effect concentration developed by McDonald, et al. (2000) (Figure B-46). According to that

study, samples exceeding this median effect concentration are likely toxic to sediment-dwelling organisms. All ΣPAH concentrations in channel sediments were below the respective threshold effects concentration developed by Swartz (1999), and fewer than 10% of all sediments, all collected nearshore, exceeded the respective threshold (Figures B-47-B-49). Overall, this analysis suggests that the invertebrate prey of juvenile salmonids inhabiting channel sediments should be at negligible risk of adverse effects from DDT, PCB, and PAH compounds. Most nearshore sediments also pose negligible risks to salmonid prey, but there are a few locations where there is some risk of toxicity, presumably mostly from PCBs.

#### 4.6 Cumulative Risks

Toxicology theory – i.e., response addition (Könemann and Pieters, 1996) – suggests that the substances of concern in this assessment (PAHs, PCBs, and DDT and its metabolites) may interact in some additive fashion to augment adverse effects to juvenile salmon, provided exposures are of sufficient magnitude and duration. In this case, virtually all channel sediments appeared to pose only negligible risks to juvenile salmonids; therefore, there should be additive risks for channel sediments only if the contaminants possess the same mode of action. Within their classes, the congeners of PAHs and PCBs and of DDT and its metabolites can be assumed to possess the same mode of action, but it cannot be assumed that PAHs have the same mode of action as PCBs or DDT, DDD or DDE<sup>15</sup>. For example, Halter and Johnson (1974), in their testing of DDT and PCB in combination using coho salmon embryos and alevins, observed much different toxicity signs and no additive toxicity; the toxicity of DDT-PCB combinations was due solely to DDT, reflecting its more rapid mode of action. The PAHs exert narcosis in acute toxicity and a variety of immunological and physiological responses in chronic toxicity that differ from chronic PCB and DDT toxicity. Therefore, as Könemann and Pieters (1996) point out for accumulating like responses to different toxicants, effects should be added only when the exposures to each of the stressors exceed their thresholds. In other words, if they do not exert an effect on their own, response addition will not occur. Nevertheless, these judgments are uncertain because they are based on the general aquatic toxicology literature rather than site-specific studies. This requires extrapolation to the exposures, life stages, and species examined here.

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<sup>15</sup> A congener is a compound having the same basic chemical structure as others in a chemical class, such as the different Aroclors of PCBs.

## 5 ASSUMPTIONS AND UNCERTAINTIES IN RISK PREDICTIONS

All of the risk predictions are based on a variety of assumptions and limited data concerning the fate and effects of the three chemical groups in aquatic ecosystems. Both the assumptions and data used in this assessment carry uncertainties that are important to consider in evaluating the risk estimates. In general, it is necessary to assume that studies conducted elsewhere can be extrapolated to the juvenile salmonid life stages, water quality, and sediment quality that typify the lower Columbia River and estuary.

### 5.1 Key Assumptions Concerning Exposure

The principal assumptions concerning exposure relate to the biota-sediment accumulation factor and the degree of exposure juvenile salmon receive in the estuary.

#### Biota-Sediment Accumulation Factor

Biota-sediment accumulation factors are known to be quite variable, and Wong et al. (2001), Watanabe and Bart (2001), and Thomann (2001) have recently reviewed the reasons for this variability. They identified the following factors as potentially influencing BSAF magnitude:

- **Assuming That a Single BSAF Applies to All PAHs and PCBs Included in The Summations:** For example, Thomann (2001) notes that the food chain contributes minimally to bioaccumulation of low molecular weight PAHs. This reflects the reduced hydrophobicity and increased lability to degradation of these compounds. Consequently, including low molecular weight PAHs in the BSAF for ΣPAH likely overestimates risk potential.
- **Biomagnification Up the Food Chain:** BSAFs do not account for biomagnification. Biomagnification may not be important at the base of food chains – for example transfer of detritus to epibenthic invertebrates (primary consumer); see Figure 3 in Suedel et al., 1994. Biomagnification of chlorinated hydrocarbons becomes more important farther up the food chain. Biomagnification potential is expected to be greatest for the most chlorinated PCBs and DDE. Any biomagnification would underestimate risk potential.
- **Actual Exposures *In Situ* Are Different Than Assumed:** It takes time (e.g., weeks) for the contaminants present at very low concentrations and having great hydrophobicity (and hence high potential BSAFs) to bioaccumulate. The longer juvenile salmon spend in the estuary, the larger the BSAF. Conversely, the shorter the time, the smaller the BSAF. One major factor affecting BSAF magnitude for each chemical and species/life stage of juvenile salmonid is the duration each stock spends in the estuary and the degree to which they eat sediment-dwelling prey rather than prey that live in the water column (i.e., plankton and nekton). Meador (2000a) notes that BSAF will increase up to a theoretical maximum with increasing residence time, all other factors being constant. Because the chemicals being considered partition slowly from food into tissue, it may take weeks to months for the maximum theoretical BSAF to be achieved when contaminant concentrations in the sediments are very low, as they certainly are in most lower Columbia River sediments. Therefore, the BSAF estimates used here (from Meador, 2000a; Wong, et al., 2001) may have overestimated risks to ESU stocks that do not feed epibenthically or spend less time in the estuary to achieve maximum transfer of contaminants from sediment to fish tissues.
- **Species and Life Stage:** Different species of juvenile salmonids have different feeding habits, and these change during their ontogeny in the estuary. Reliance on epibenthic prey decreases as fish

grow, and some species do not prey significantly on epibenthic invertebrates at any time. The BSAF should decrease proportionately to the degree juvenile salmonids feed on fish or pelagic invertebrates. In this report, it was assumed that juvenile salmon fed exclusively on epibenthic prey, thus likely overestimating risk potential (Watanabe and Bart, 2001).

- **Type of Organic Carbon:** The chemicals considered here have different affinities for different types of organic carbon, and the carbon types present in the lower Columbia River have not been compared to the ones used to derive BSAFs. It is unknown whether this factor will increase or decrease risk.
- **Equilibrium Between Contaminant Concentrations in The Different Environmental Phases (e.g., Sediment or Tissue):** Steady-state BSAFs are based on the assumption that tissue residues are in equilibrium with the sediment. In a dynamic (tidally driven) environment like the Columbia River estuary, this assumption appears unlikely, and therefore BSAF may be overestimated.

Overall, the assumptions above tend to be conservative and should overestimate risks.

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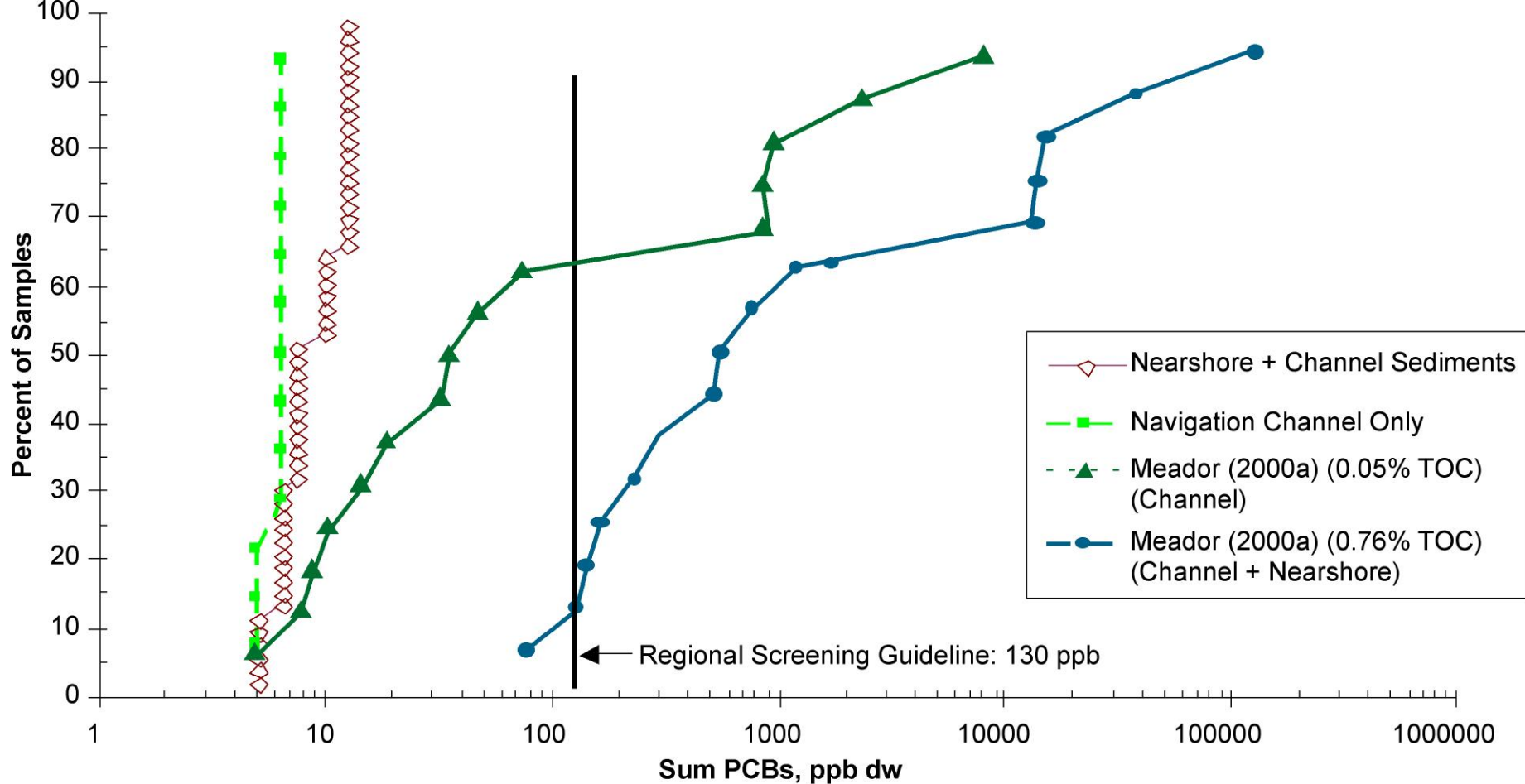
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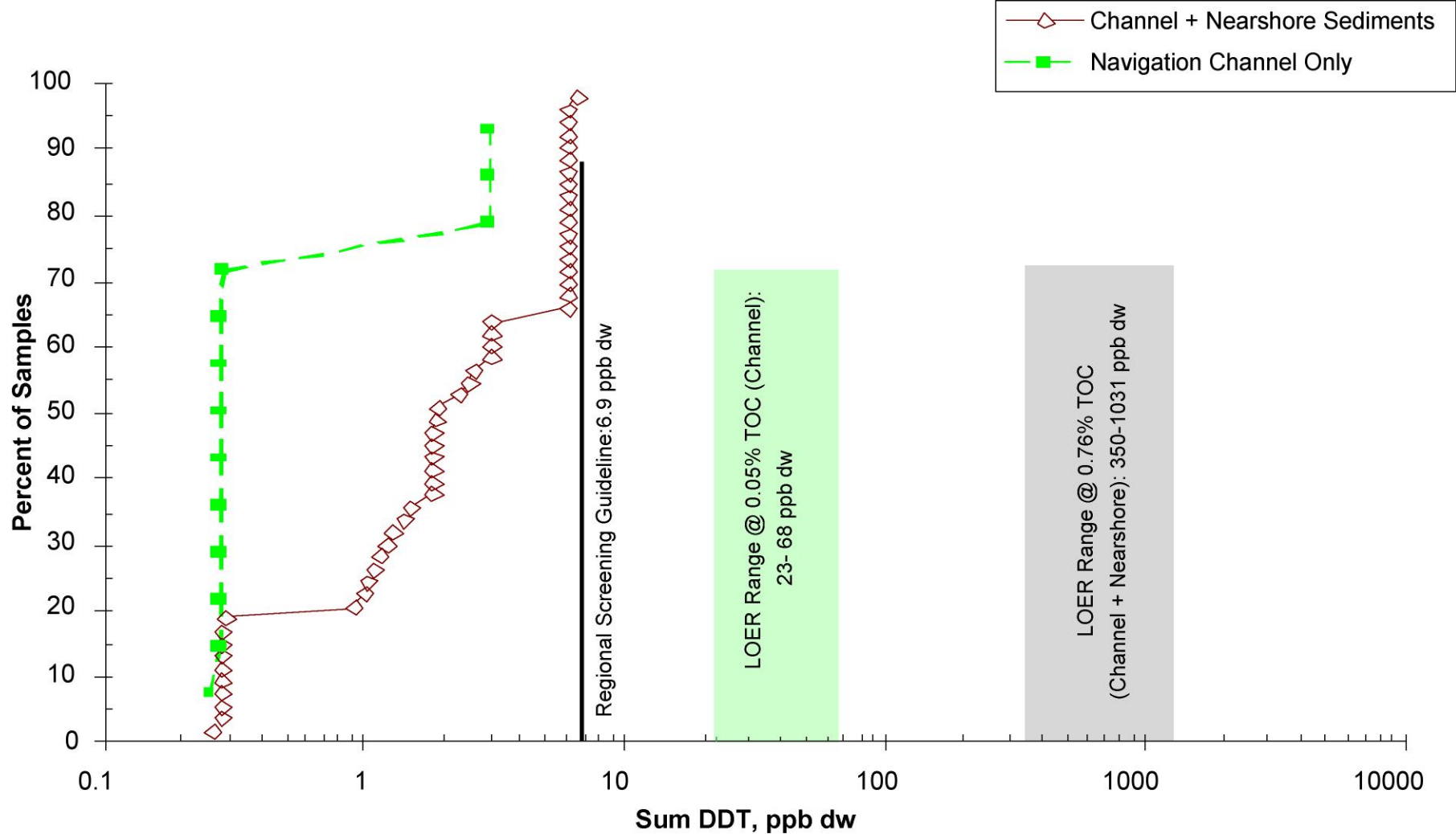
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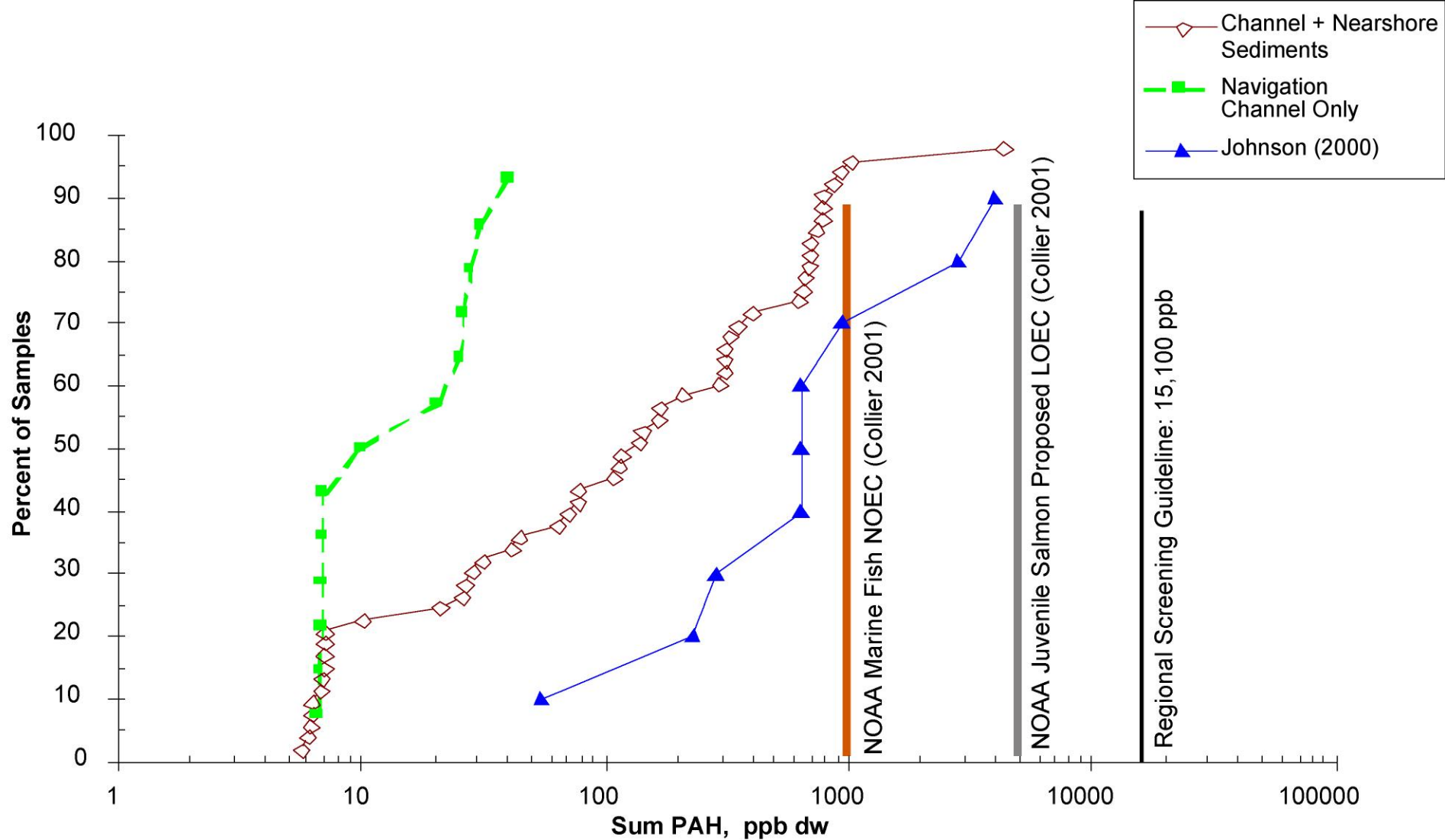


Source: Meador 2000a

**Figure B-1**  
**Concentrations of PCBs in Sediments Compared**  
**to Those Associated with Adverse Effects in Fish:**  
**River Miles 0-40**



**Figure B-2**  
**Concentrations of DDT and Metabolites in Sediments**  
**Versus All Sampling Sites: River Miles 0-40**



**Figure B-3**  
**Concentrations of PAHs in Sediments Compared to Those Associated with Adverse Effects in Fish:**  
**River Miles 0-40**

Navigation Channel

PAH-TOC

Dredging

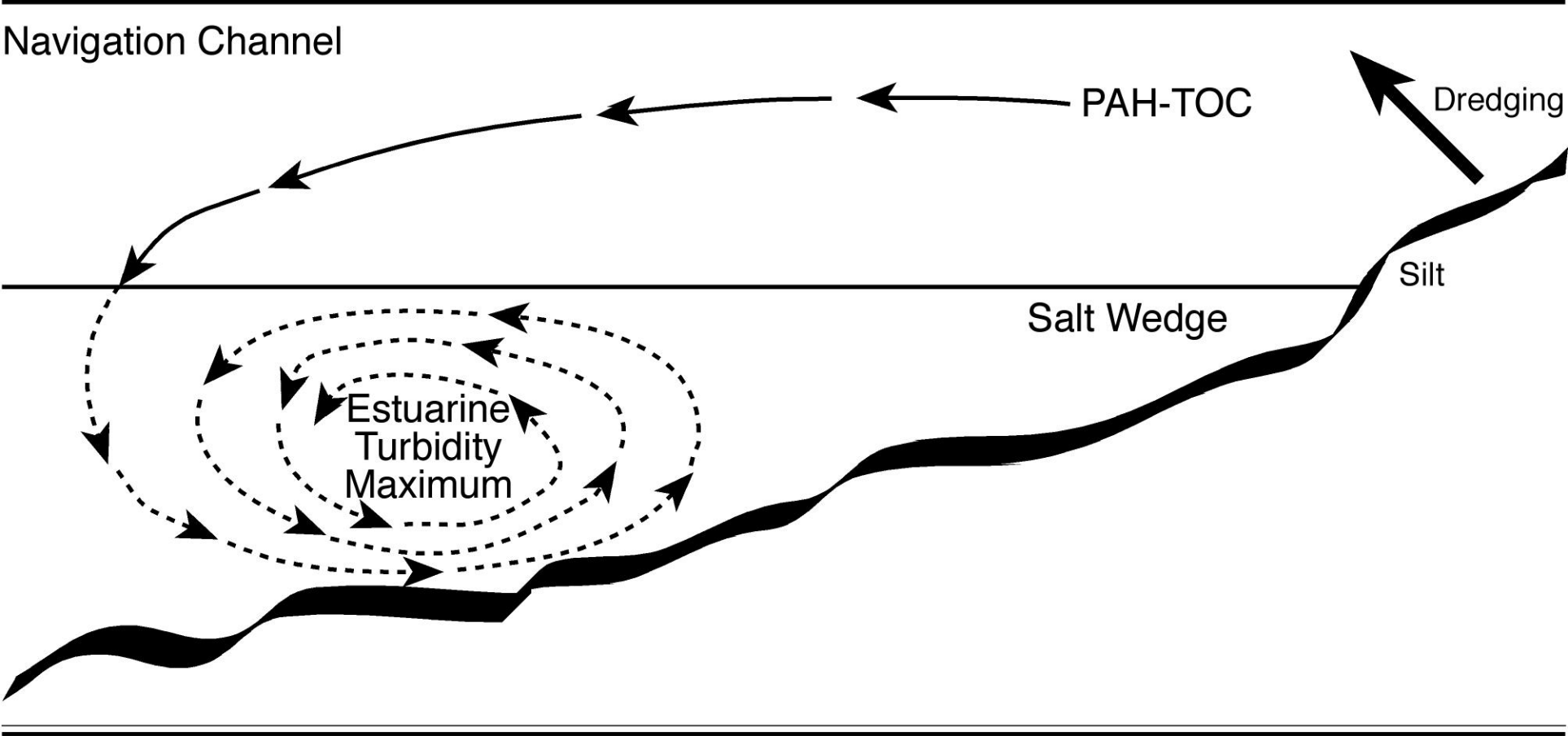
Silt

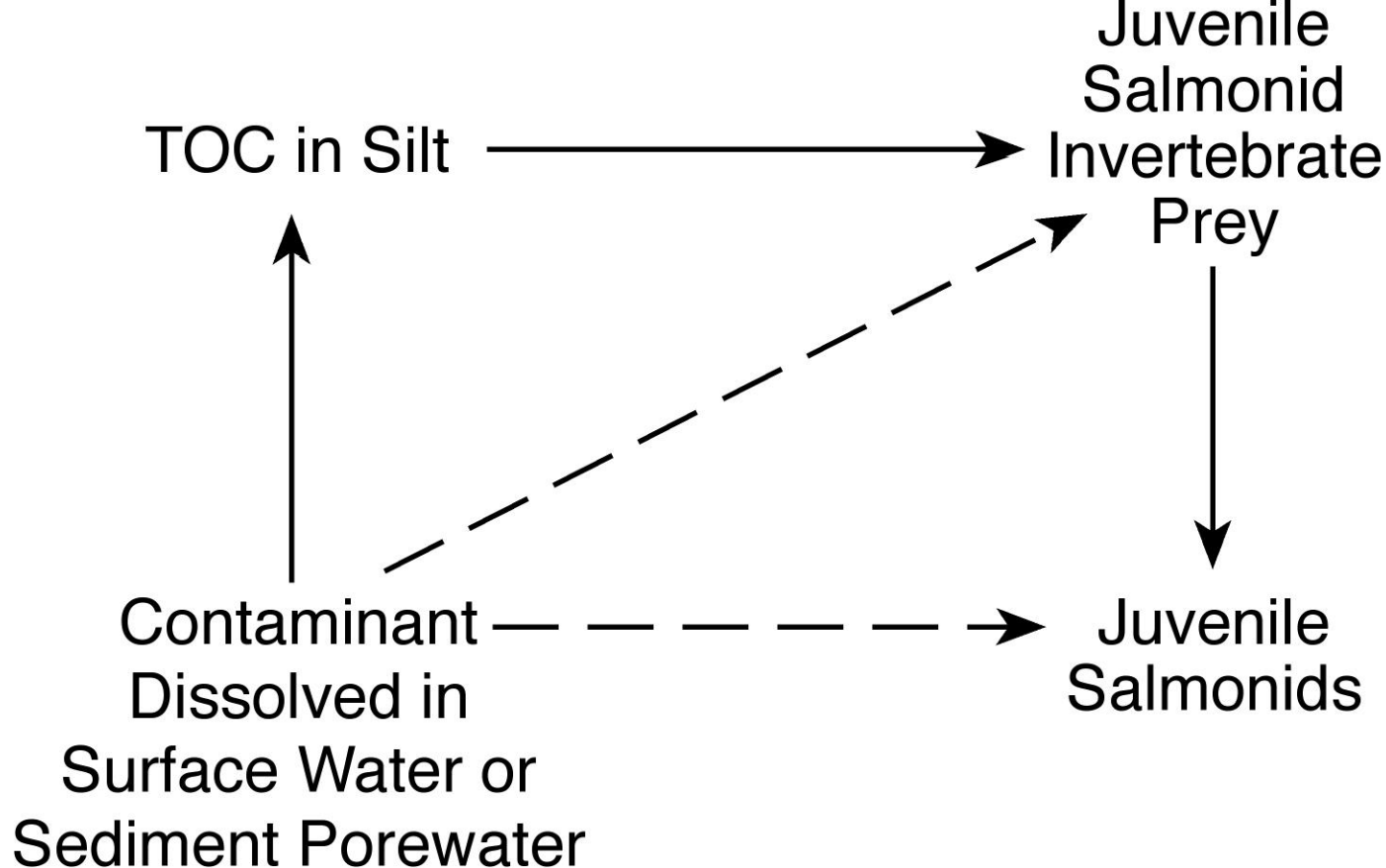
Salt Wedge

Estuarine  
Turbidity  
Maximum

Figure B-4

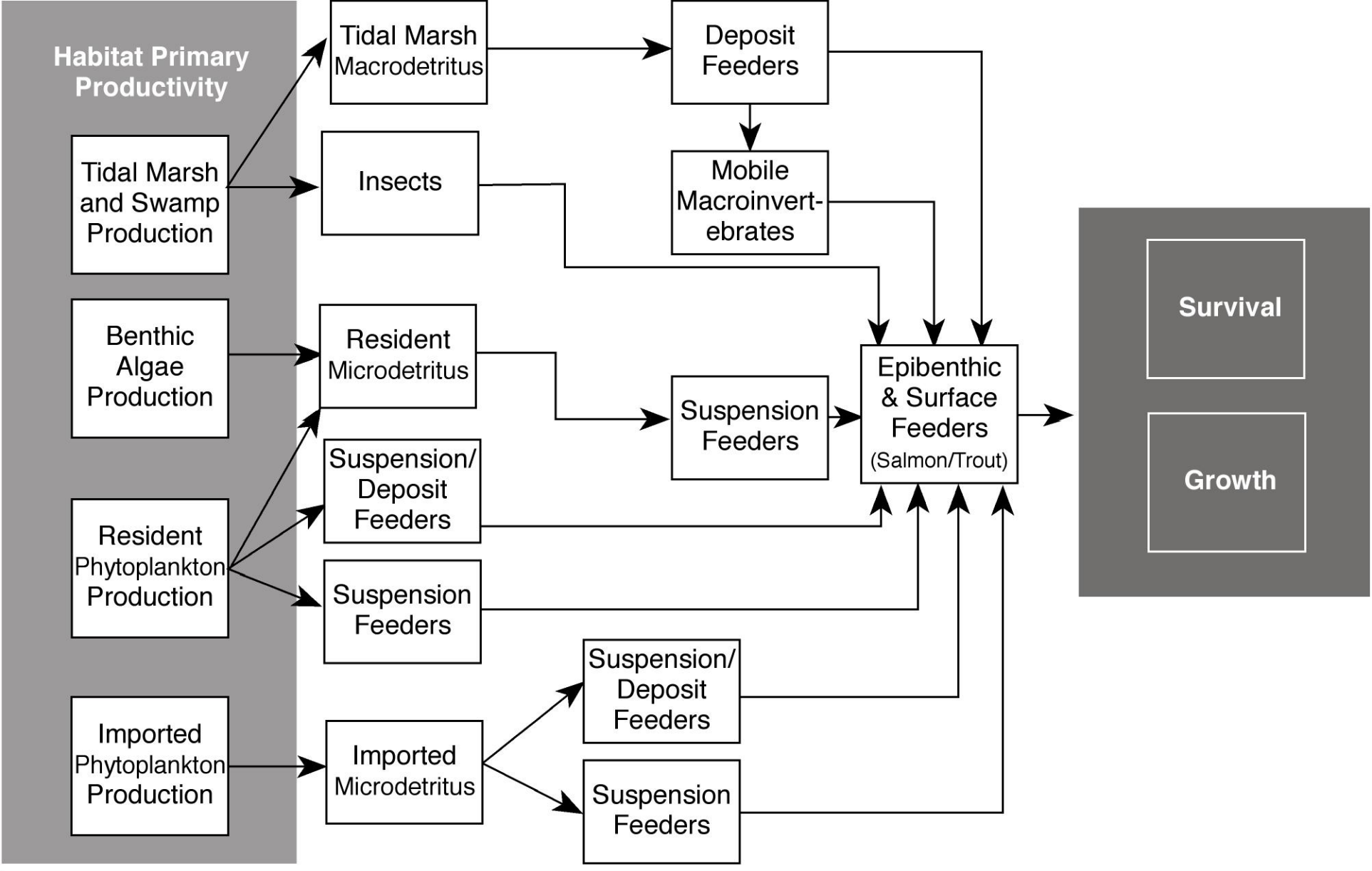
Hypothesized Relationships Between Dredging Activities in the Main Columbia River Navigation Channel, Suspension of Detritus, and Reflux of Detritus in the Estuarine Turbidity Maximum Zone





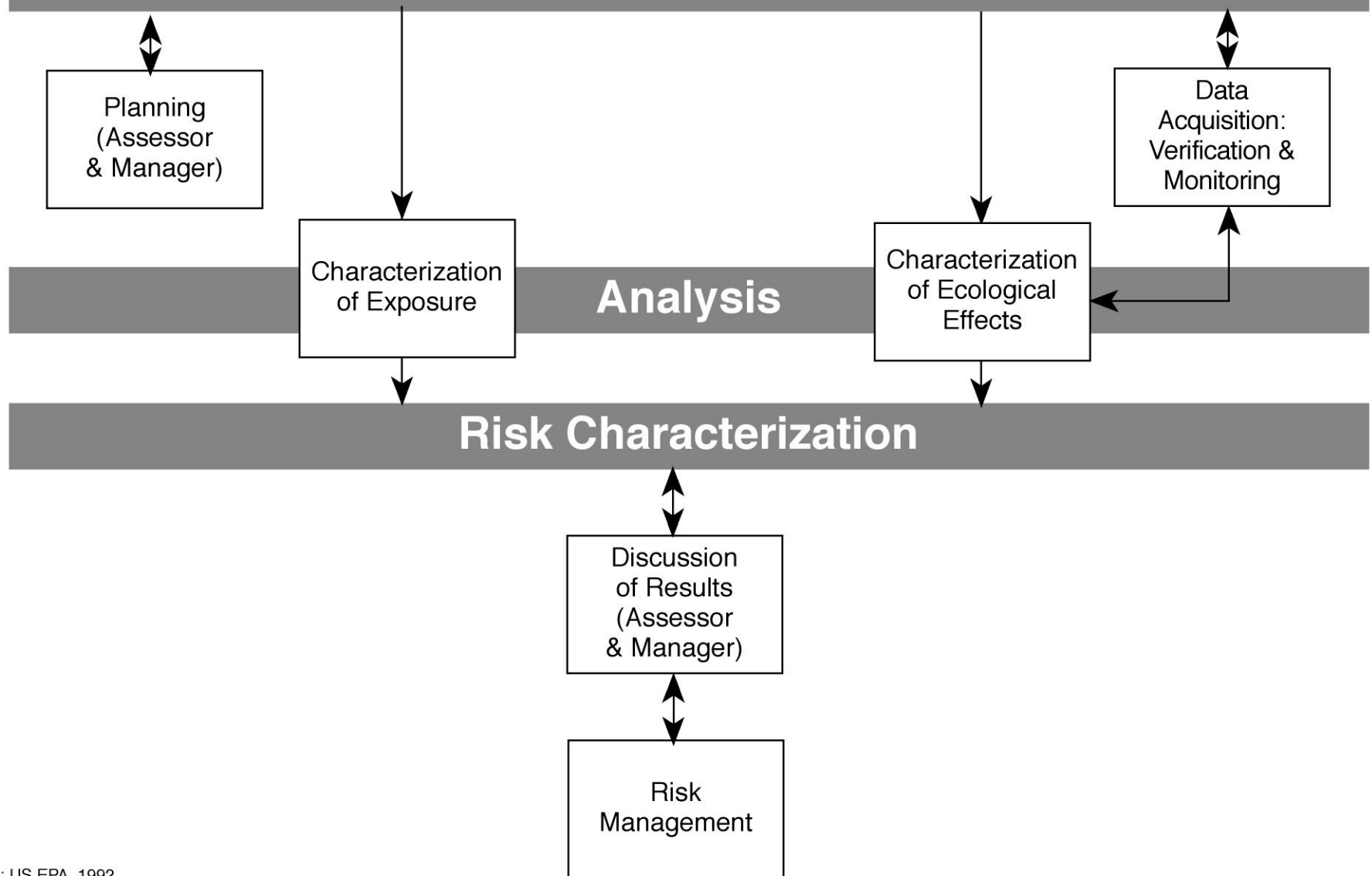
Note: Solid lines are assumed dominant pathways  
and dashed lines are assumed to be subordinant

**Figure B-5**  
**Pathways by Which Salmonids will be**  
**Exposed to Chemicals Like PCBs, DDT and**  
**Metabolites, and PAHs**



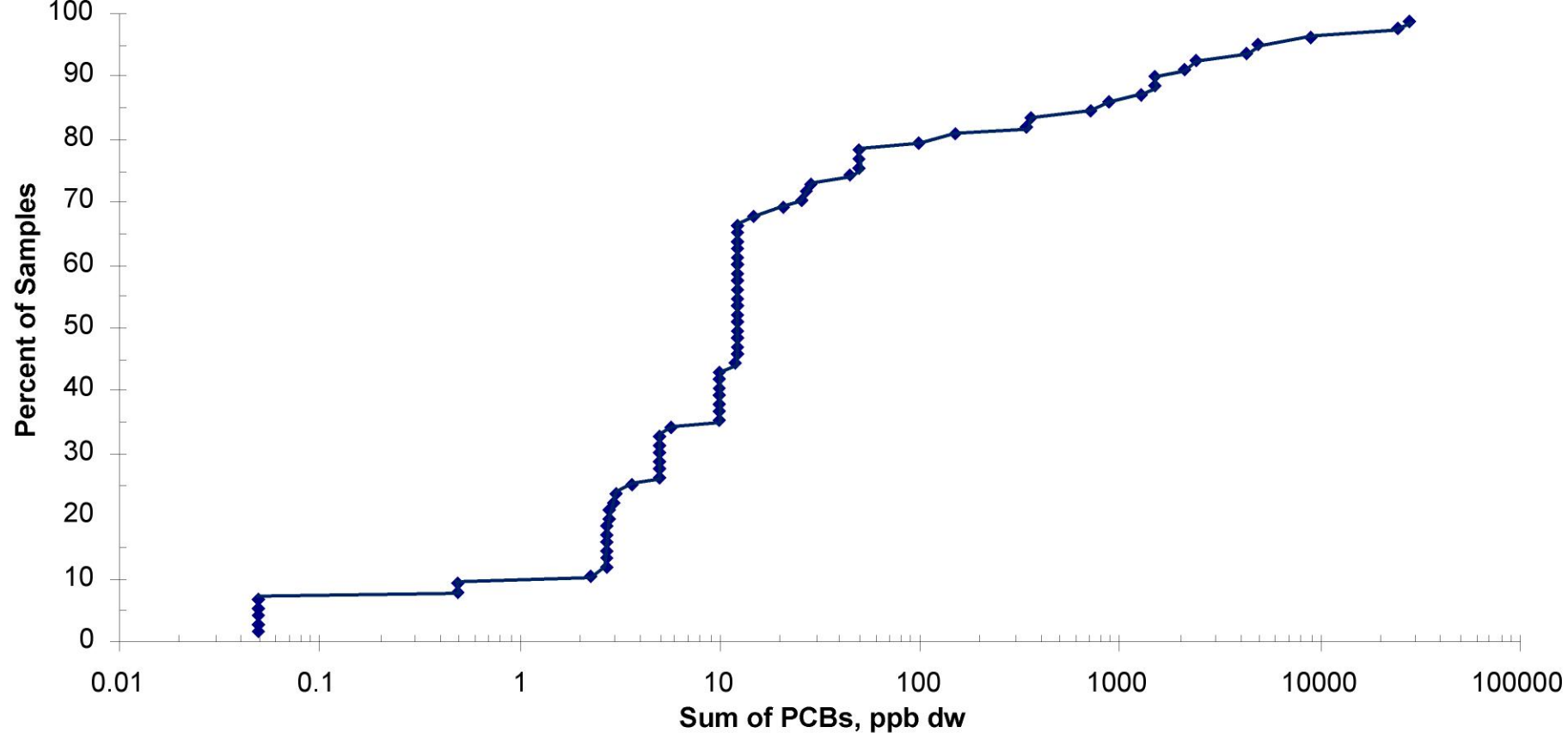
**Figure B-6**  
**Food Web and Pathways**  
**for Juvenile Salmonids**

# Problem Formulation



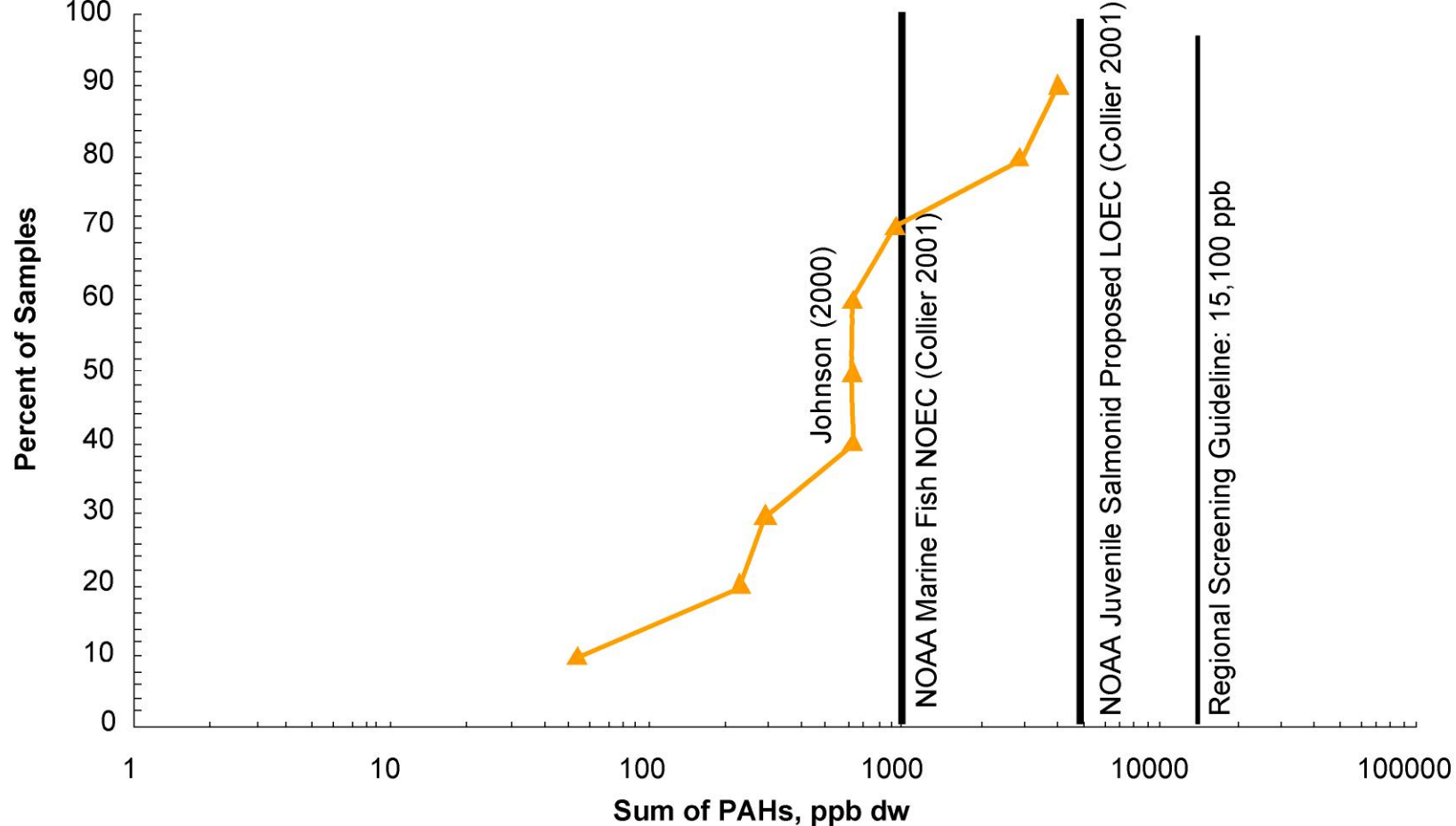
Source: US EPA, 1992

**Figure B-7**  
**Framework for Ecological**  
**Risk Assessment**

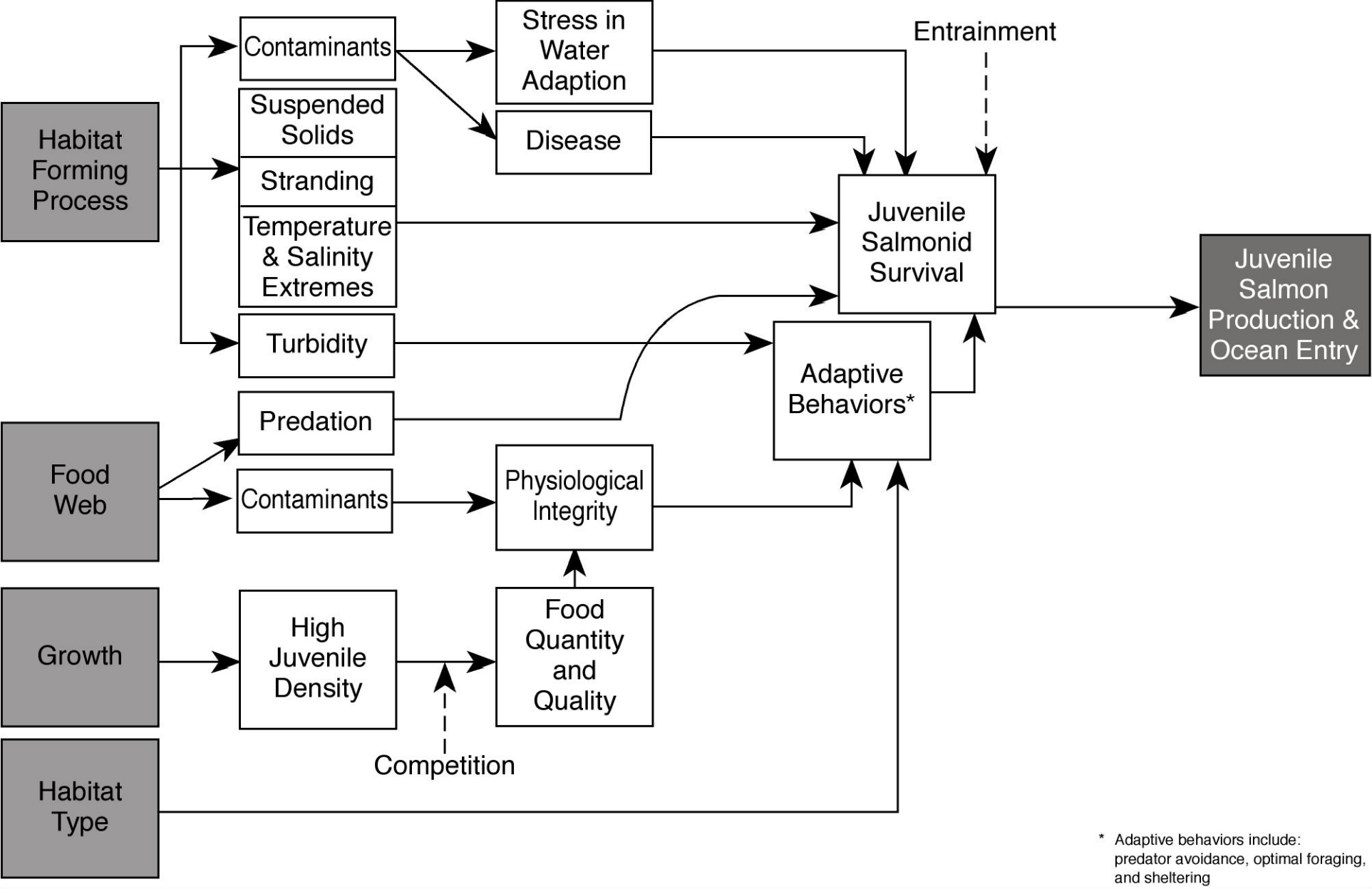


**Figure B-8**

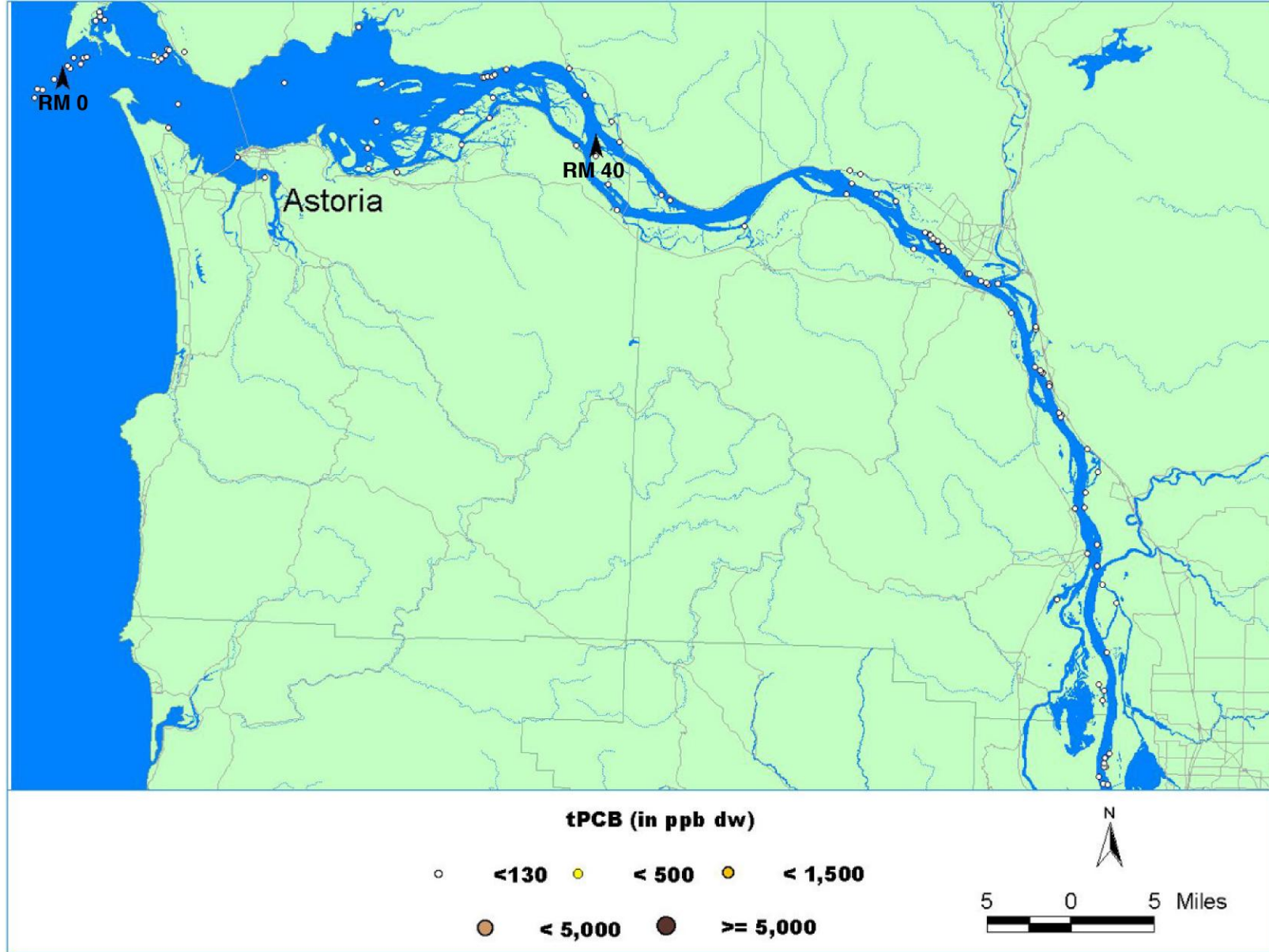
**How Exposure was Characterized for Contaminants in Sediment Using Cumulative Probability Distributions Relating Each Contaminant's Concentration to its Frequency of Occurrence in Sampled Sediments**



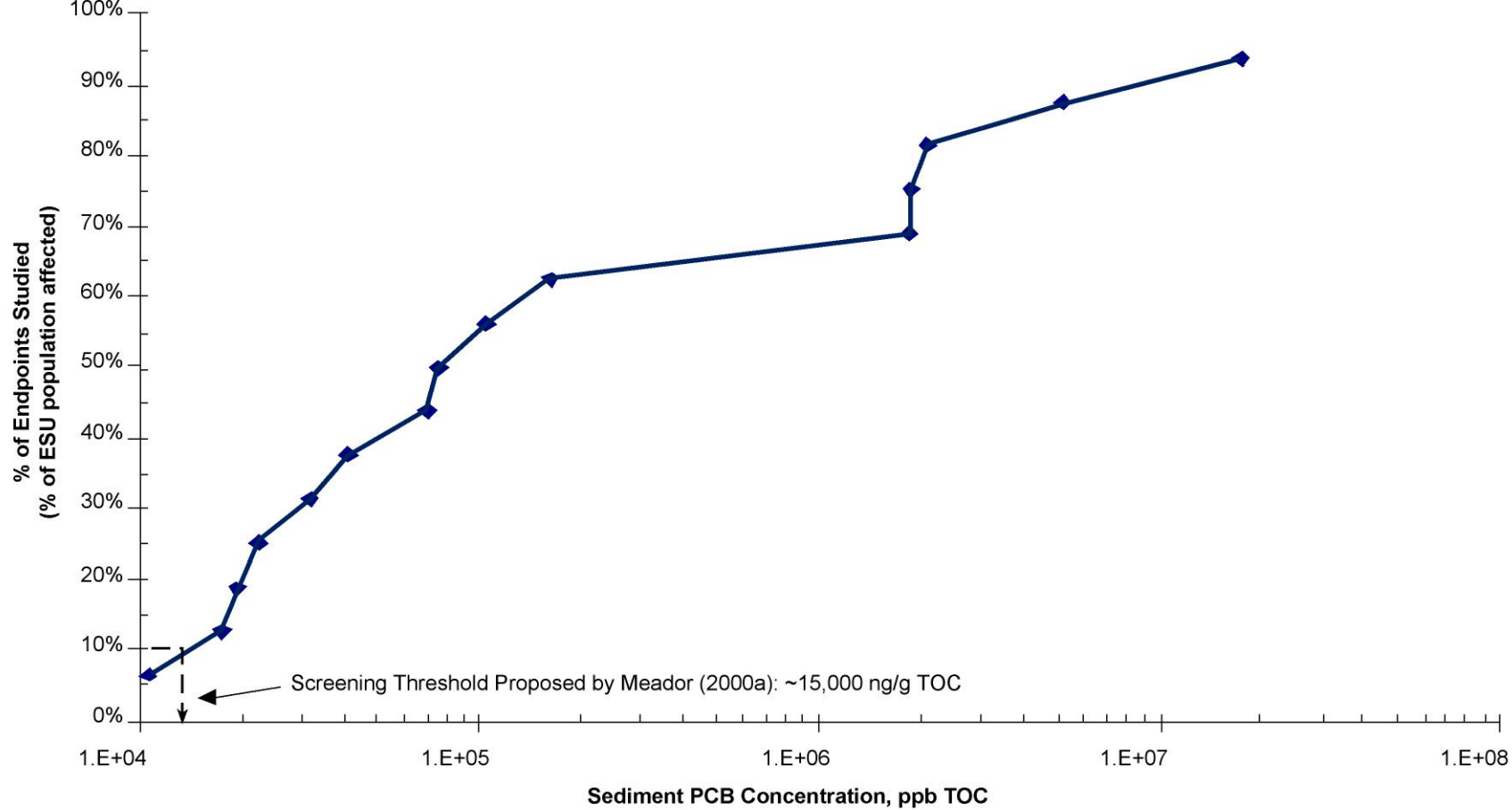
**Figure B-9**  
How Effects were Characterized Using Cumulative Probability Distributions that Related Contaminant Concentrations to Expected Effects or to Threshold Concentrations, Above Which Adverse Effects May be Anticipated



**Figure B-10**  
**Survival Pathways**  
**for Juvenile Salmonids**

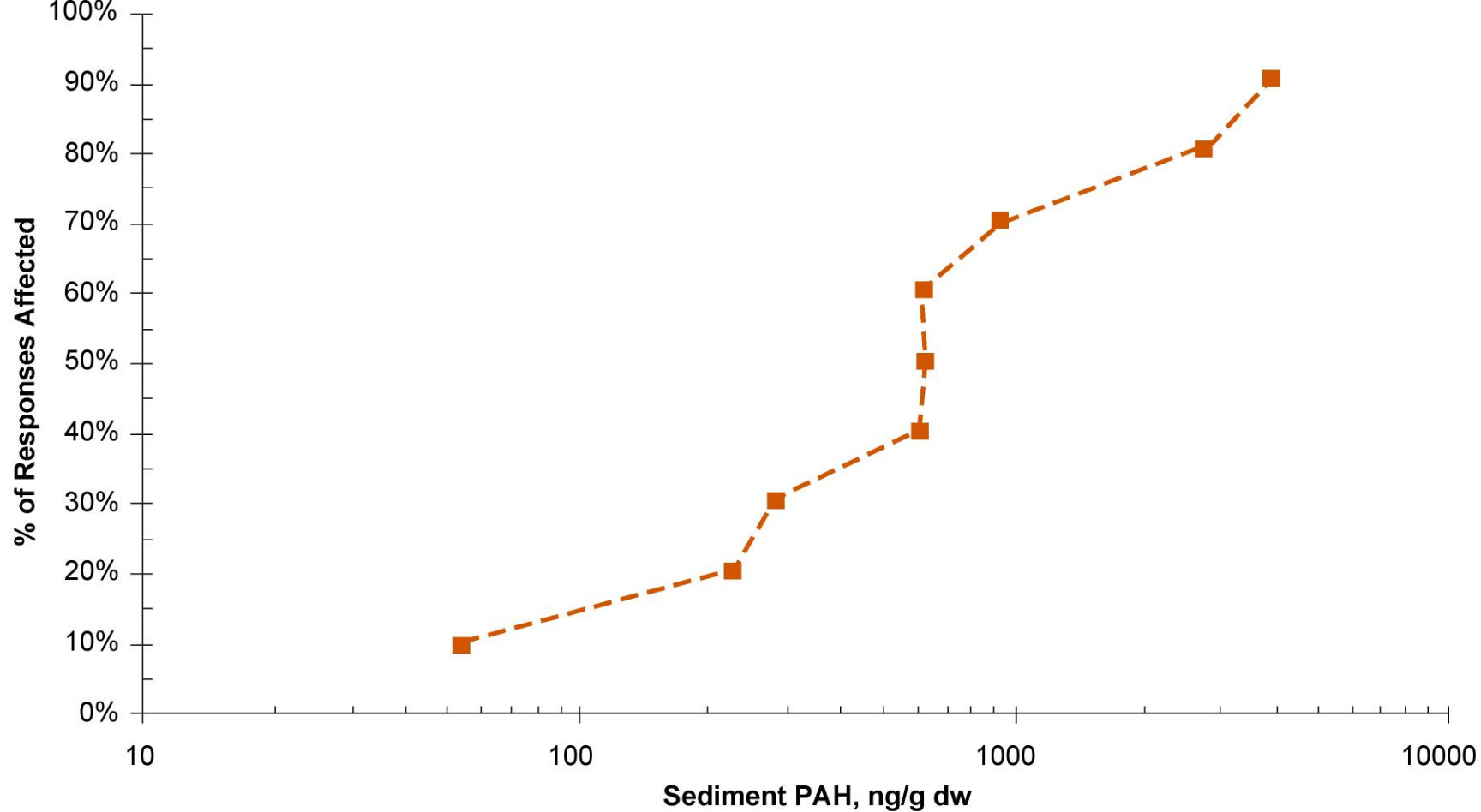


**Figure B-11**  
**Spatial Distribution of  $\Sigma$ PCB Concentrations**  
**from the Mouth of the Columbia to Willamette**  
**River Confluence**



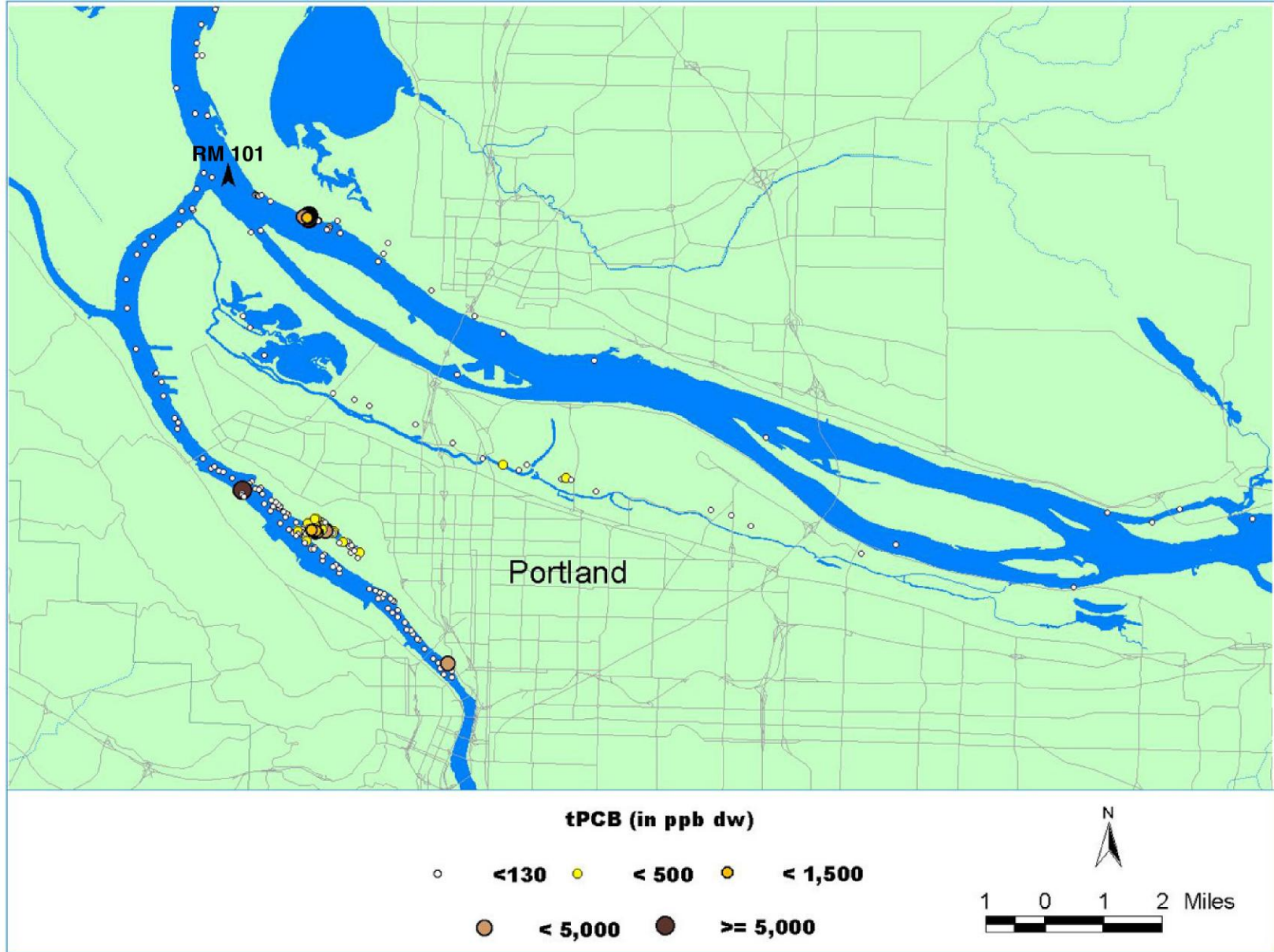
Source: Meador (2000a)

**Figure B-12**  
**Concentration-Response Relationship for**  
**Salmonids Exposed to PCBs in Their Diet**

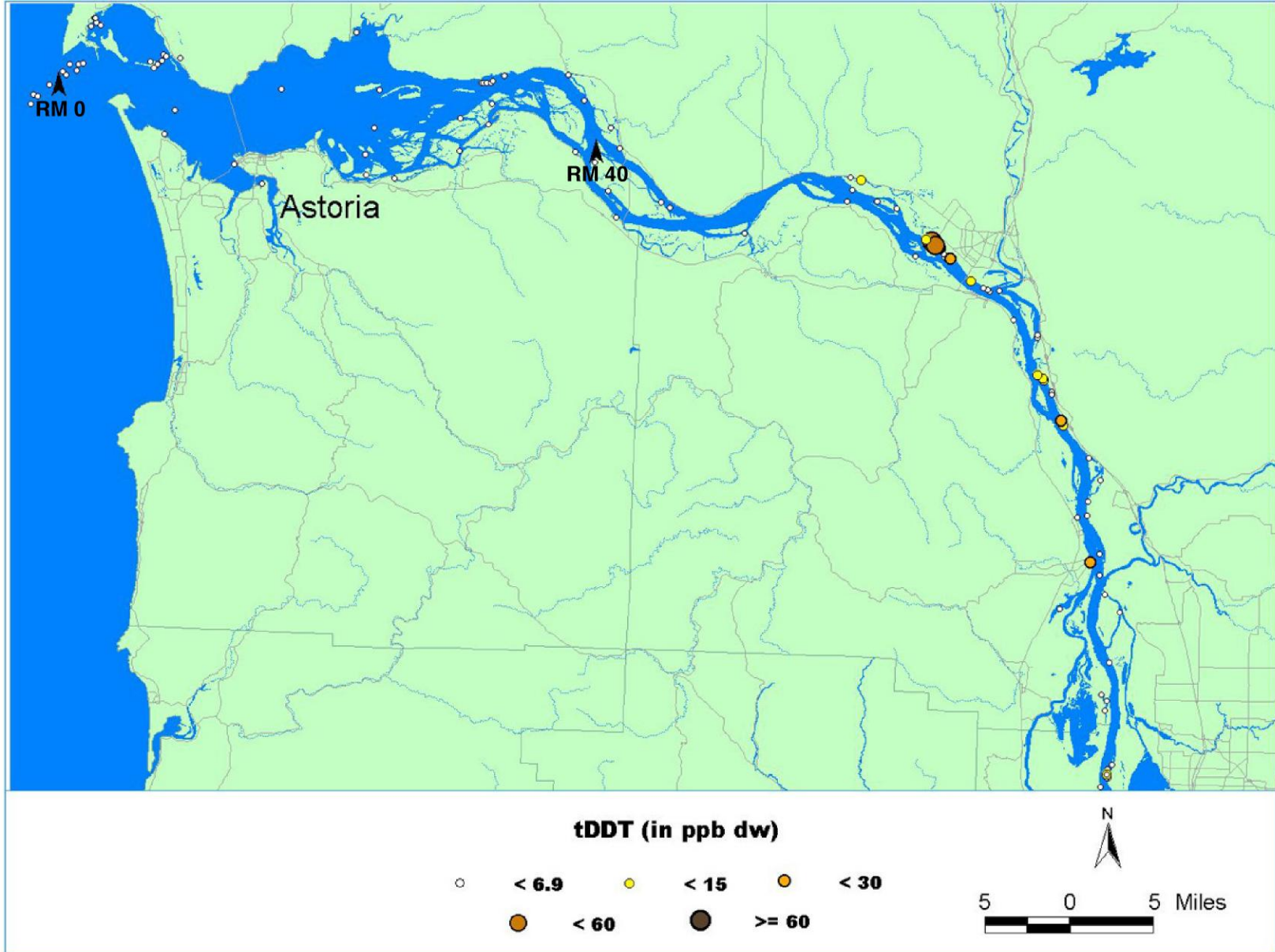


Source: Johnson, 2000

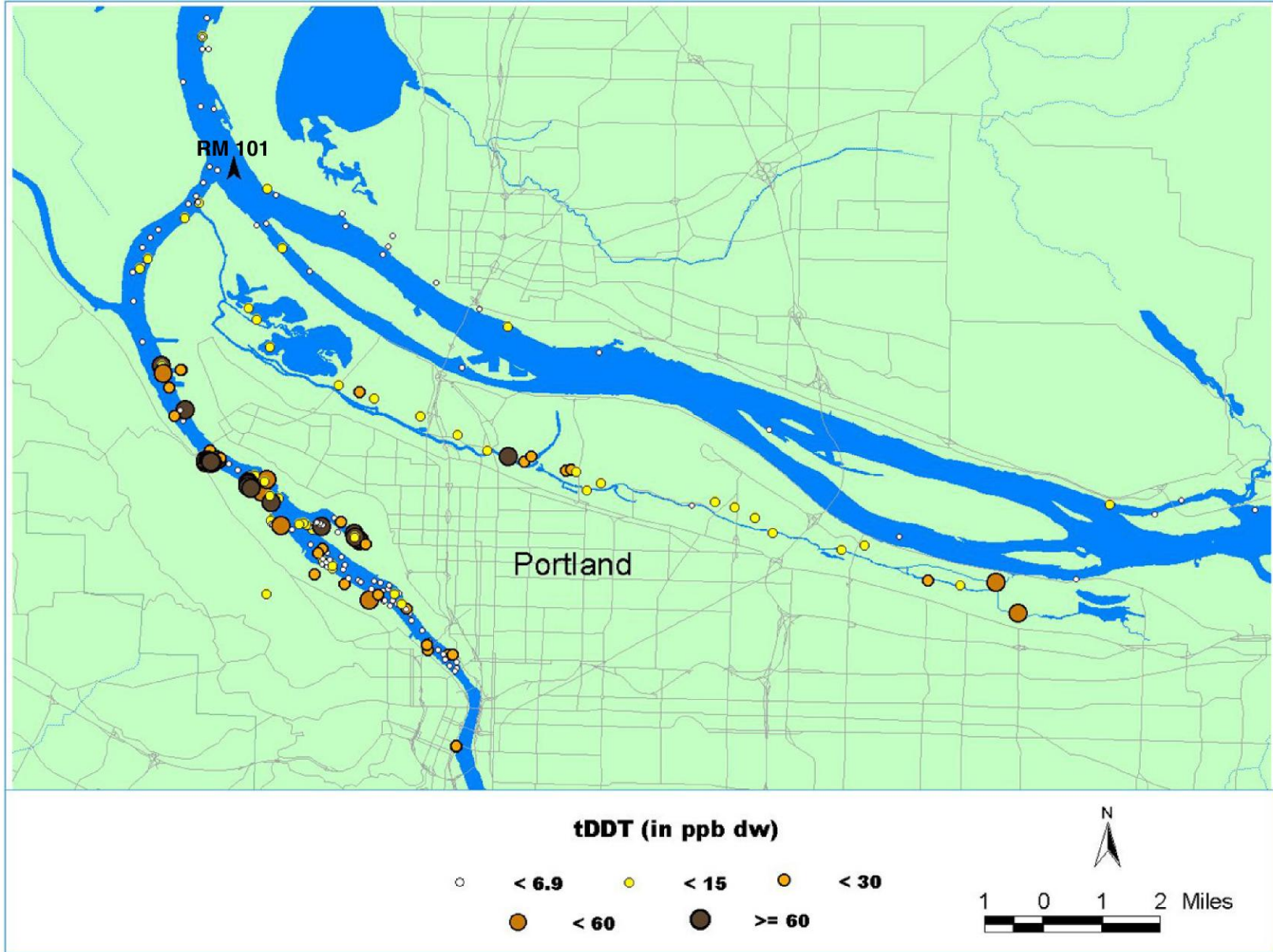
**Figure B-13**  
**Distribution of Responses in >2-Year-Old English**  
**Sole that were Correlated with Sediment**  
**Concentrations of 18 PAHs in Puget Sound**



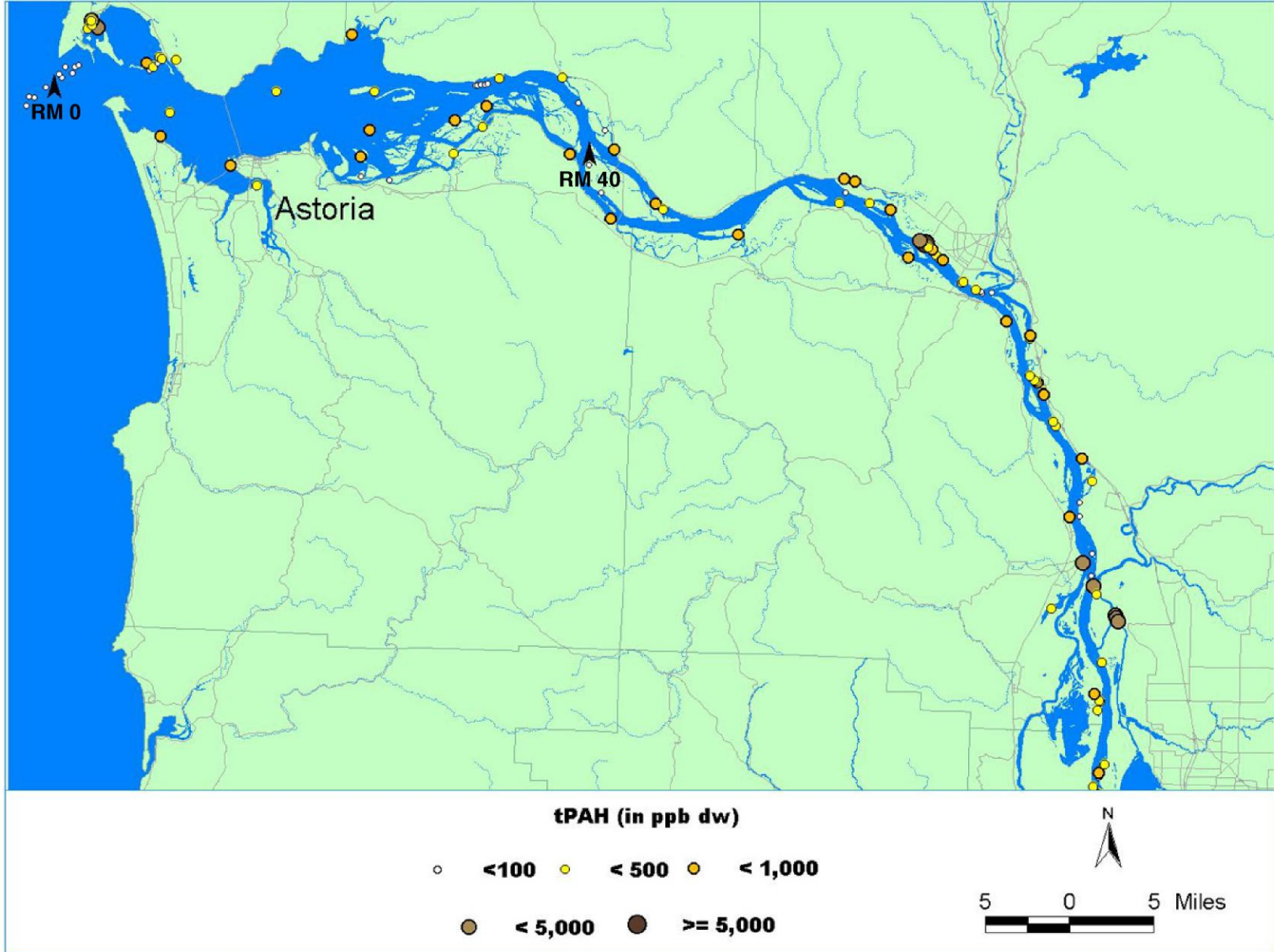
**Figure B-14**  
**Spatial Distribution of  $\Sigma$ PCB**  
**Concentrations Near Willamette-**  
**Columbia Confluence**



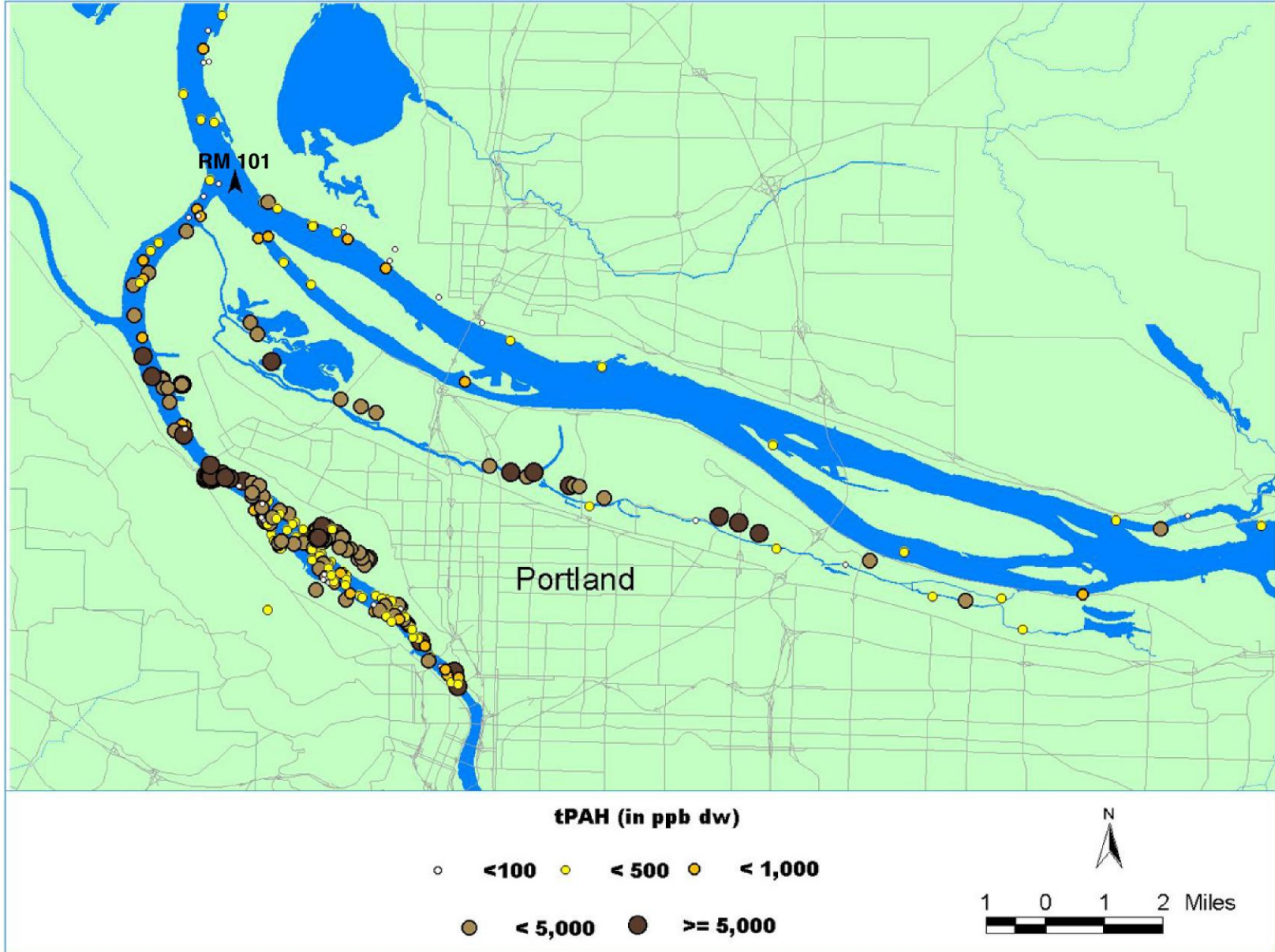
**Figure B-15**  
**Spatial Distribution of  $\Sigma$ DDT Concentrations**  
**From Mouth to Willamette River Confluence**



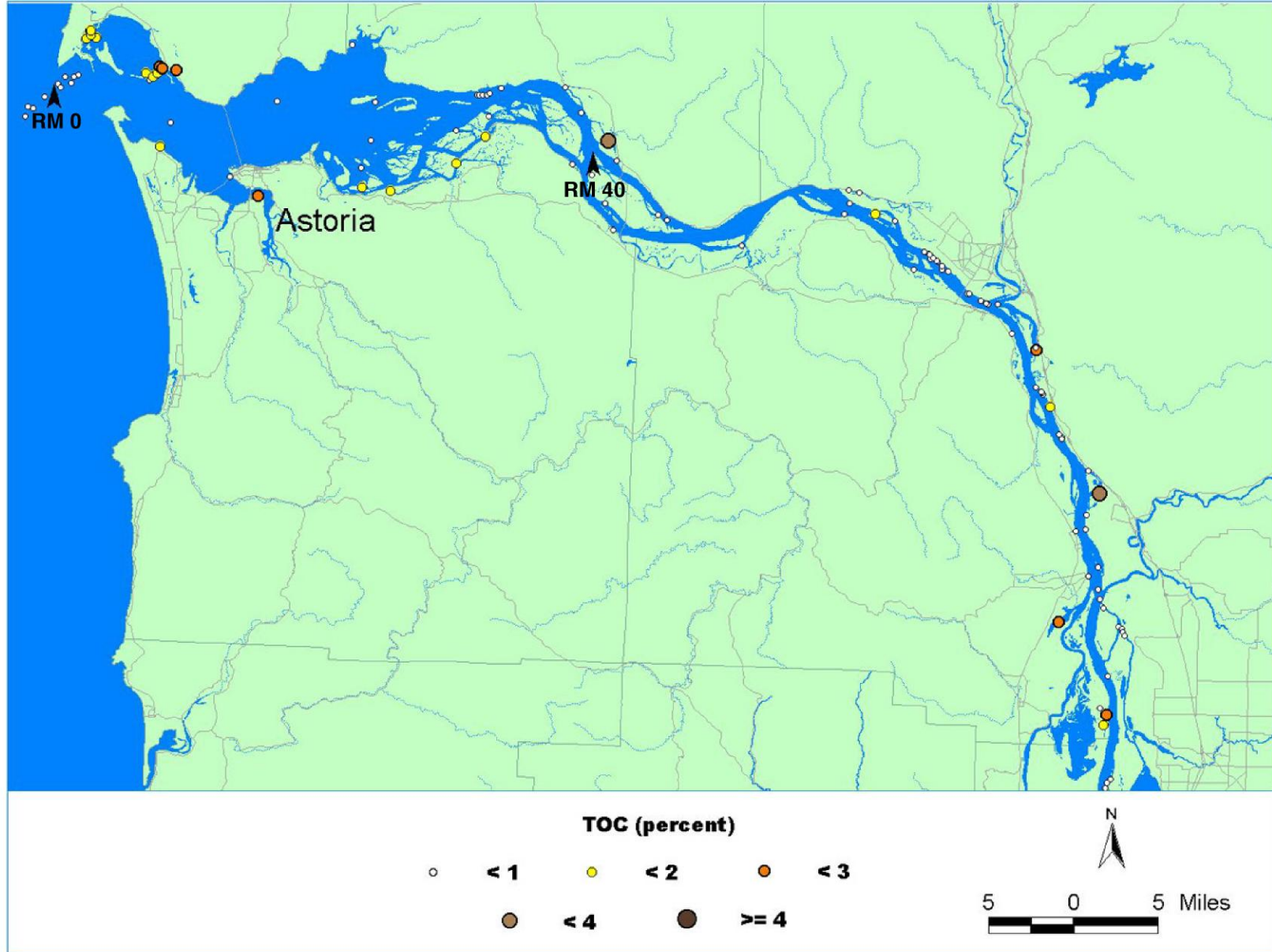
**Figure B-16**  
**Spatial Distribution of  $\Sigma$ DDT**  
**Concentrations Near Willamette-**  
**Columbia Confluence**



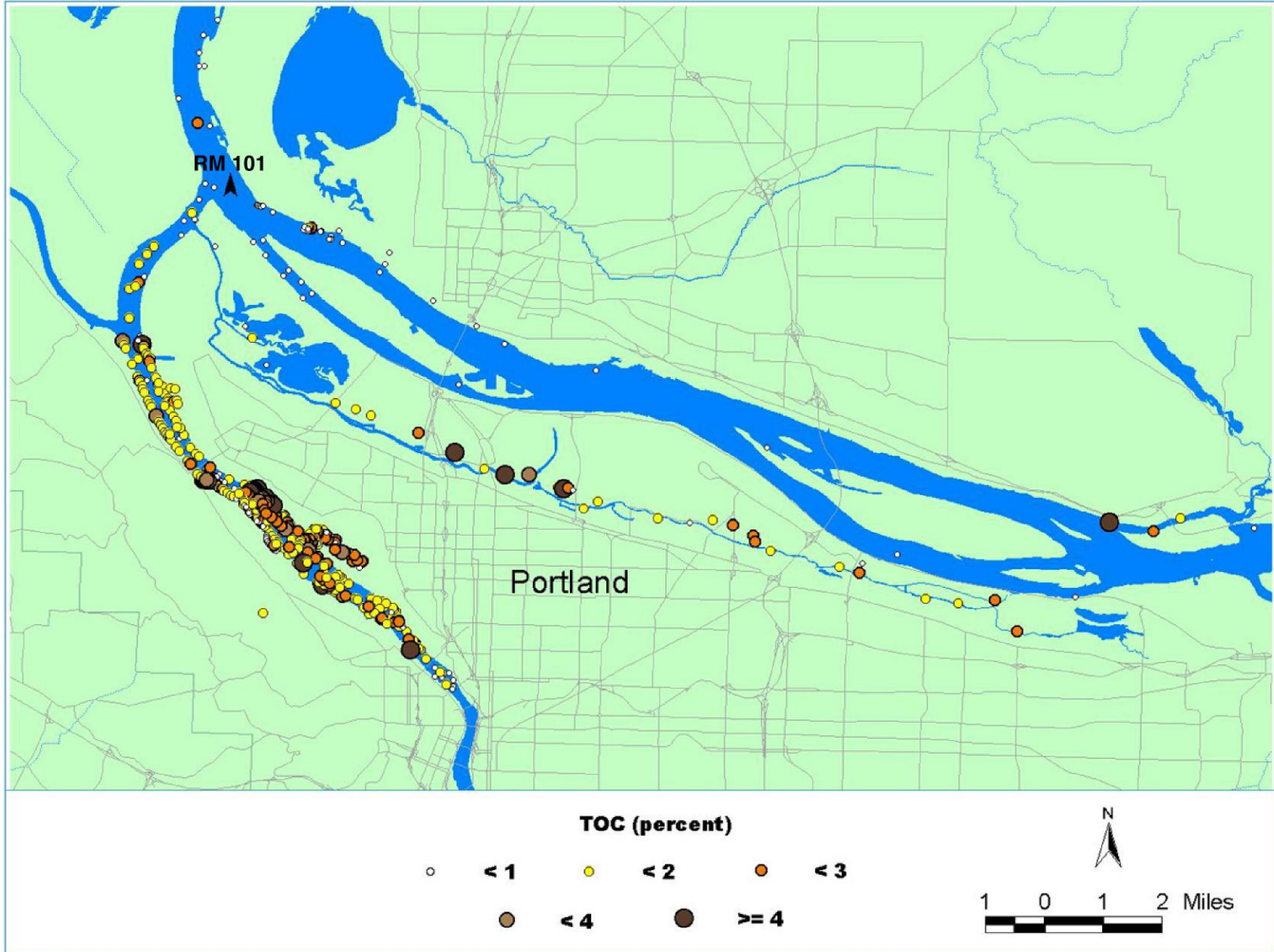
**Figure B-17**  
**Spatial Distribution of  $\Sigma$ PAH Concentrations**  
**From Mouth to Willamette River Confluence**



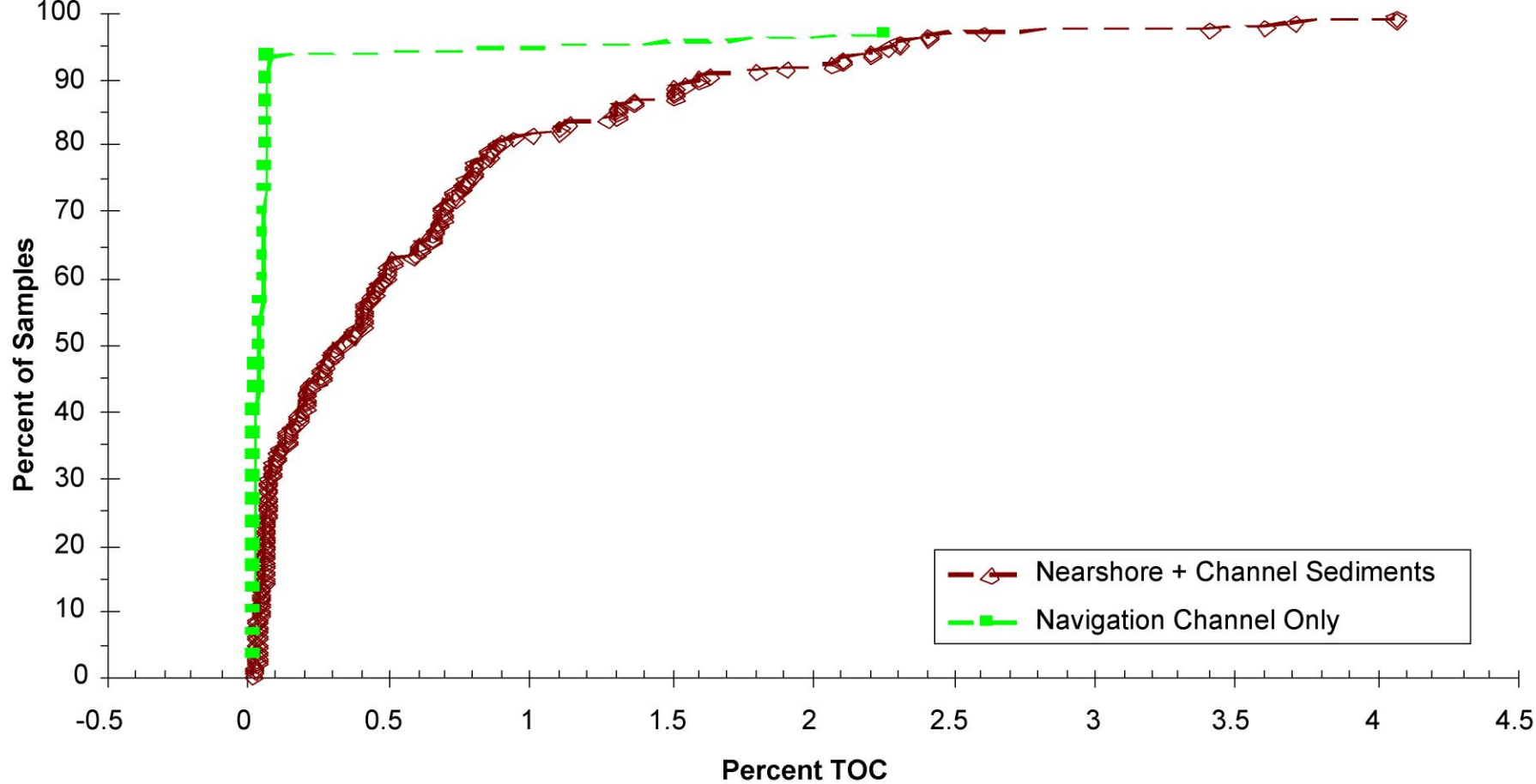
**Figure B-18**  
**Spatial Distribution of  $\Sigma$ PAH**  
**Concentrations Near Willamette-**  
**Columbia Confluence**



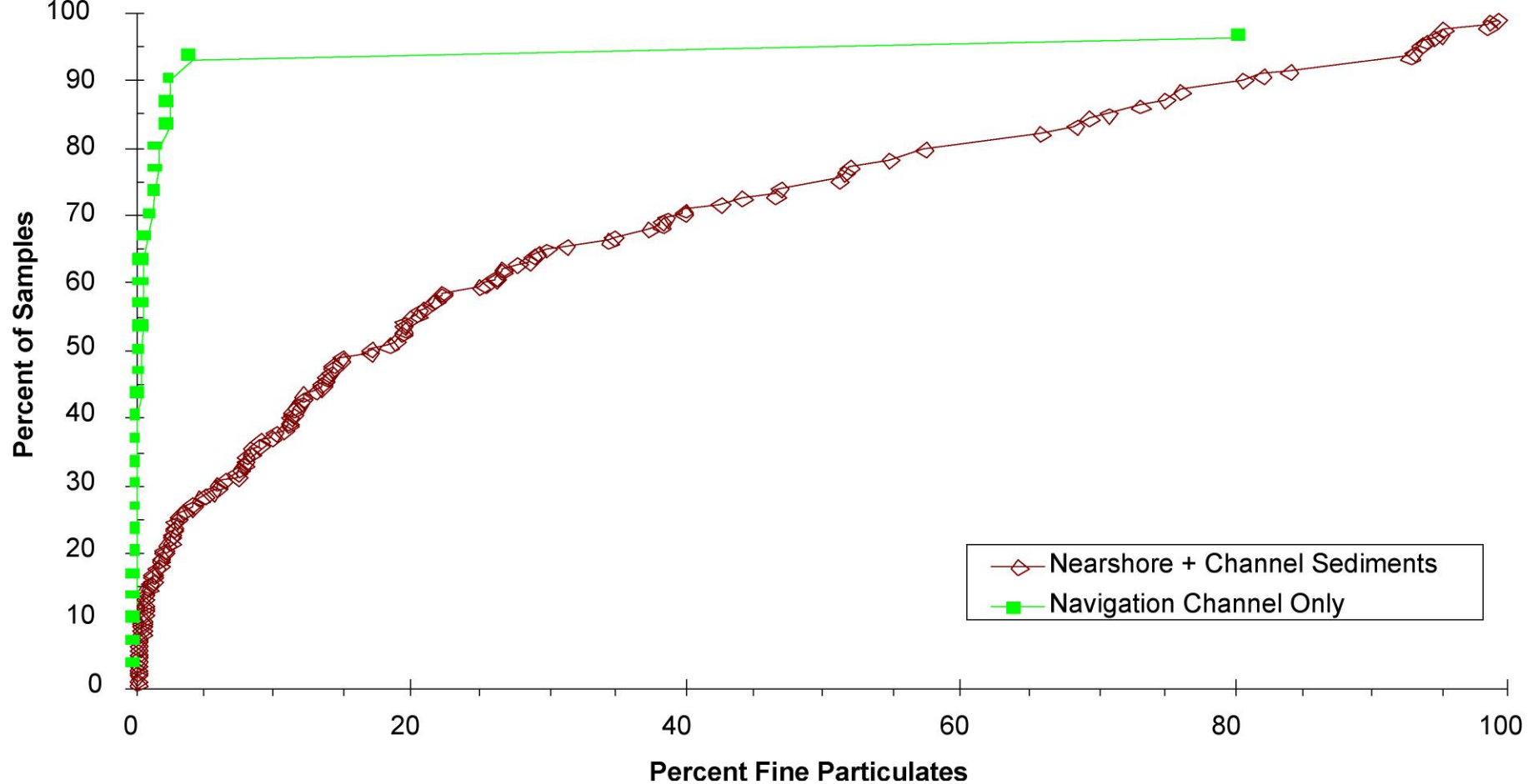
**Figure B-19**  
**Spatial Distribution of TOC Concentrations**  
**from Mouth to Willamette River Confluence**



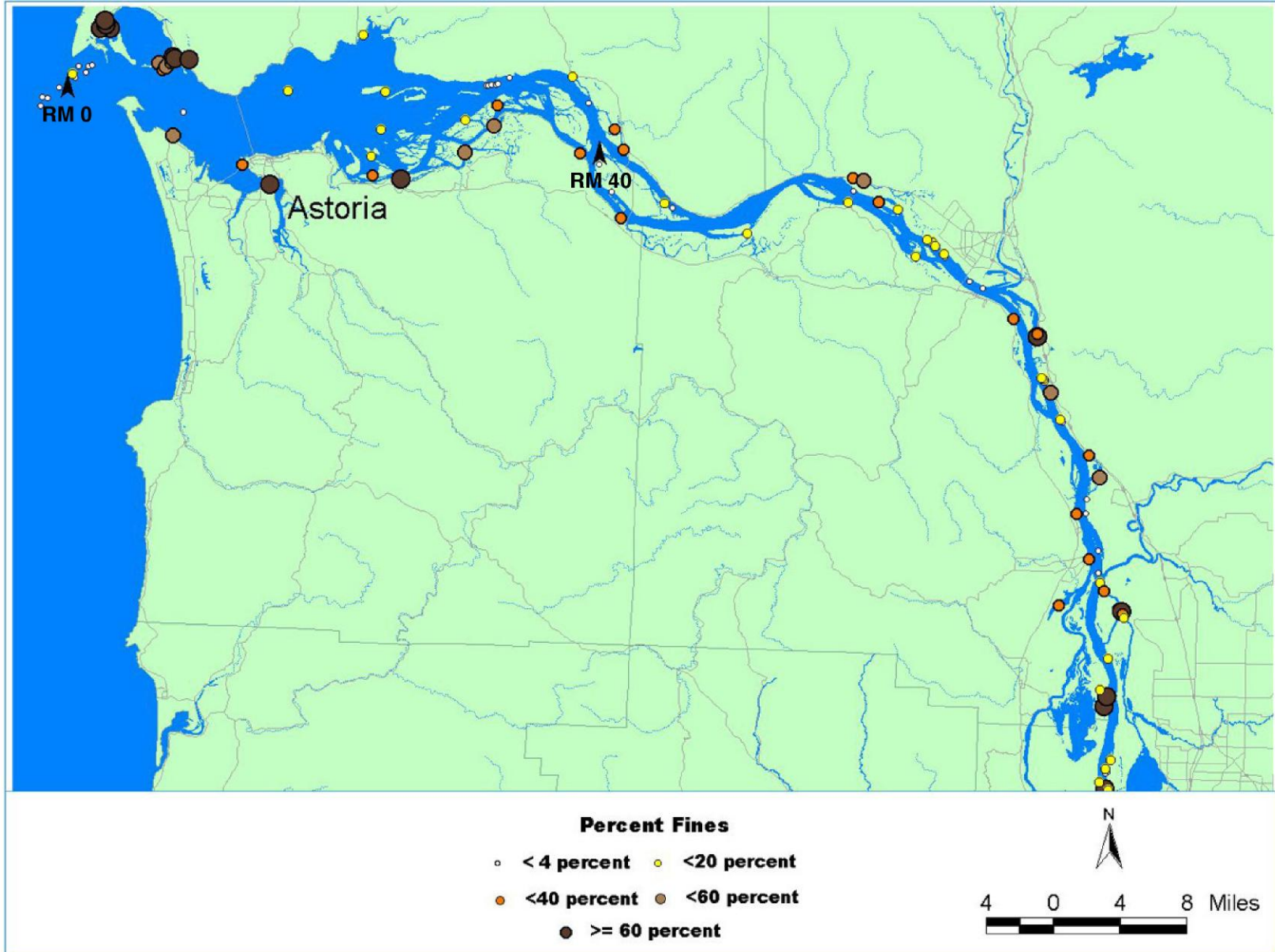
**Figure B-20**  
**Spatial Distribution of TOC**  
**Concentrations Near Willamette-**  
**Columbia Confluence**



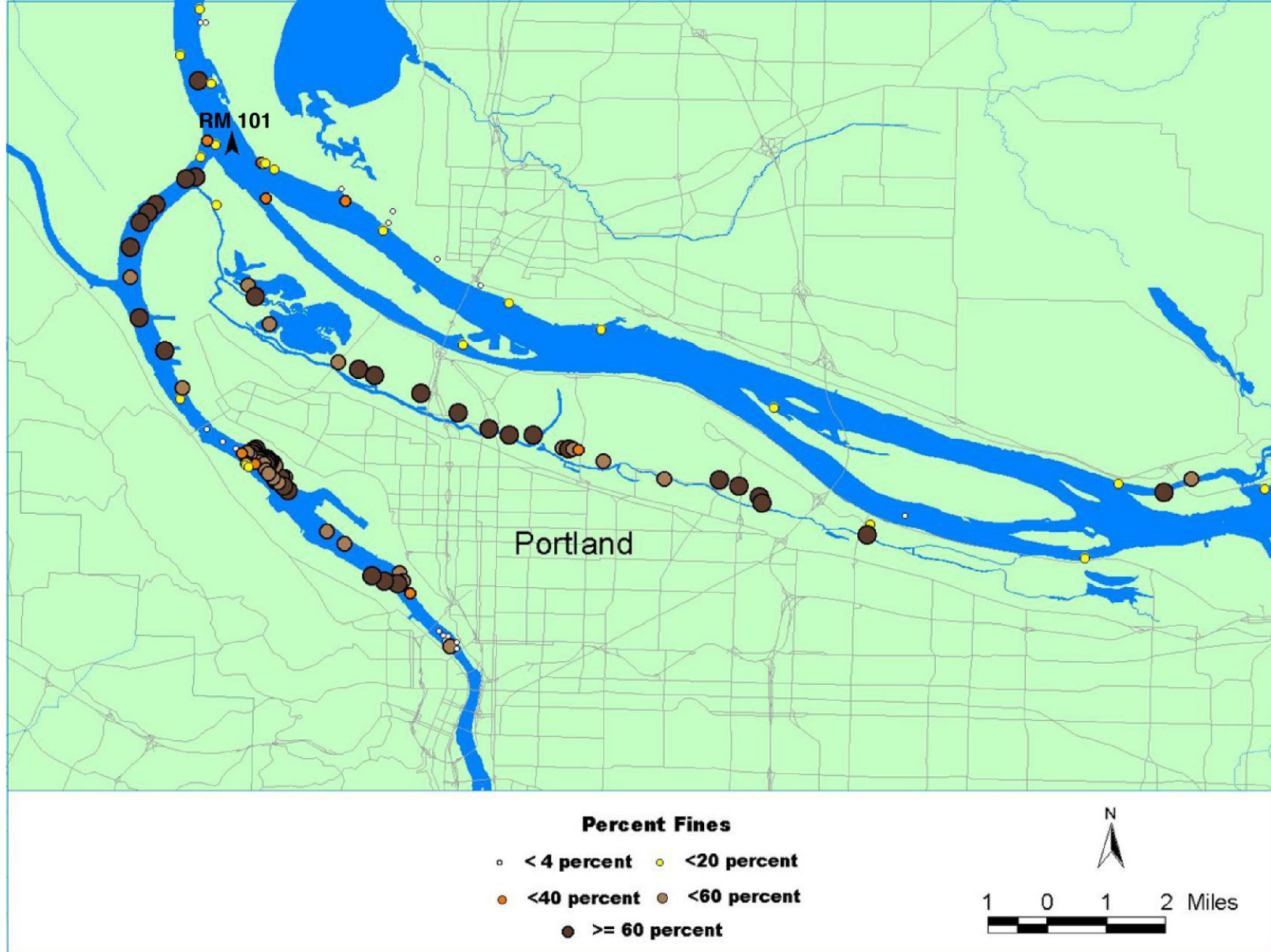
**Figure B-21**  
**Total Organic Carbon (TOC) in**  
**Columbia River Sediments (RM 0-145)**



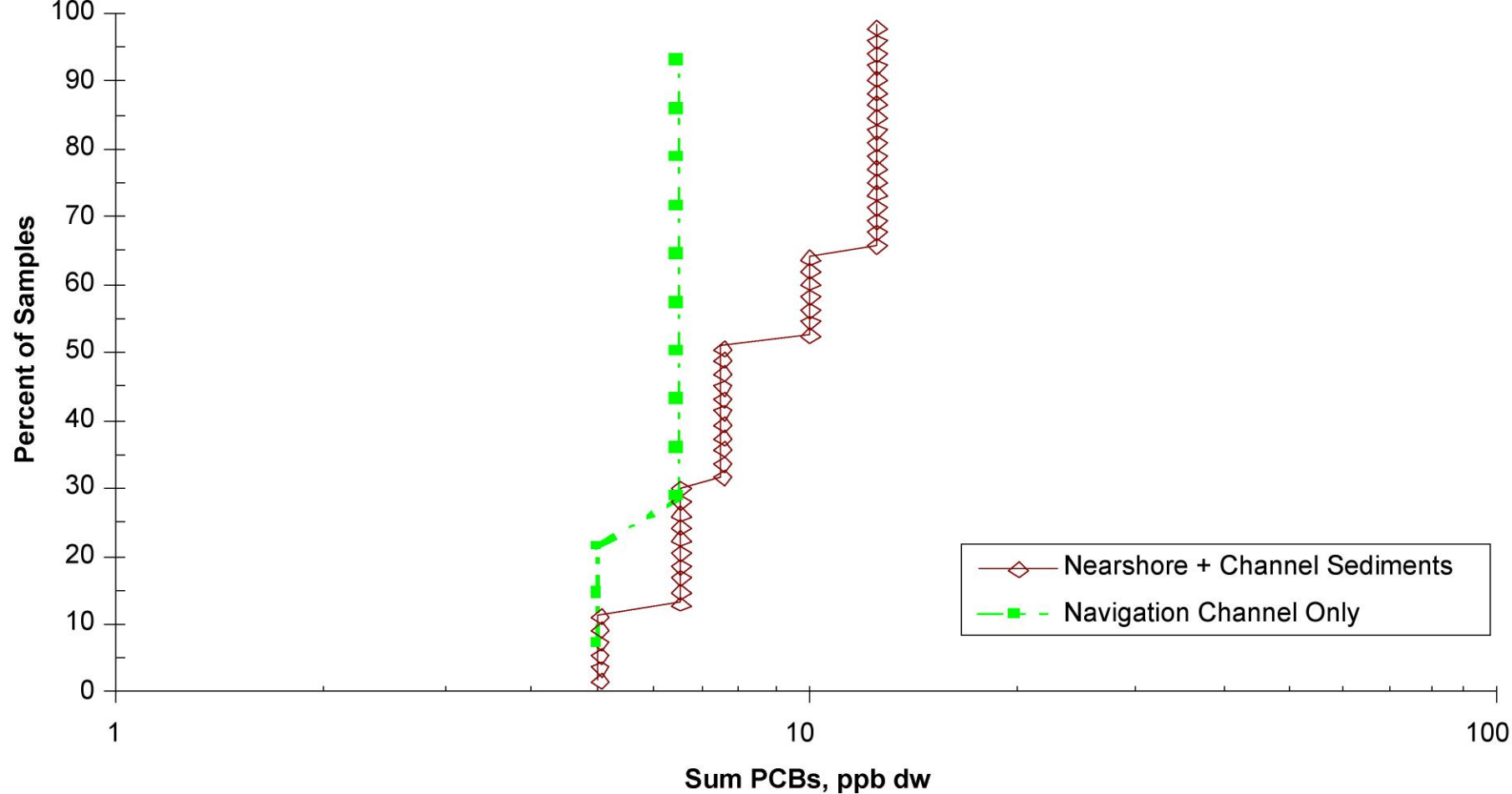
**Figure B-22**  
**Fine Particulates in Columbia River**  
**Sediments (RM 0-145)**



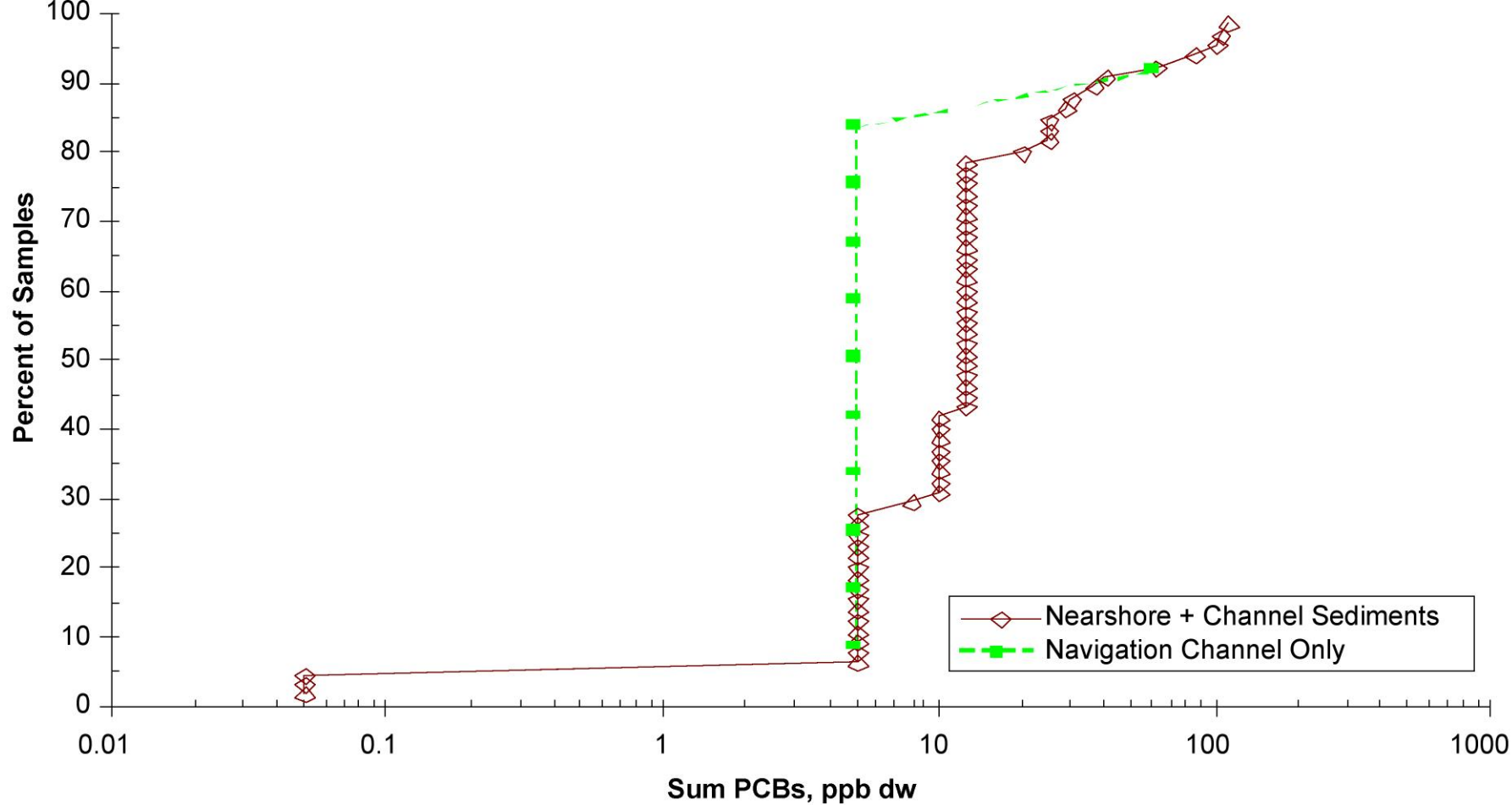
**Figure B-23**  
**Spatial Distribution of Fine Particulate Concentrations**  
**from the Mouth to Willamette River Confluence**



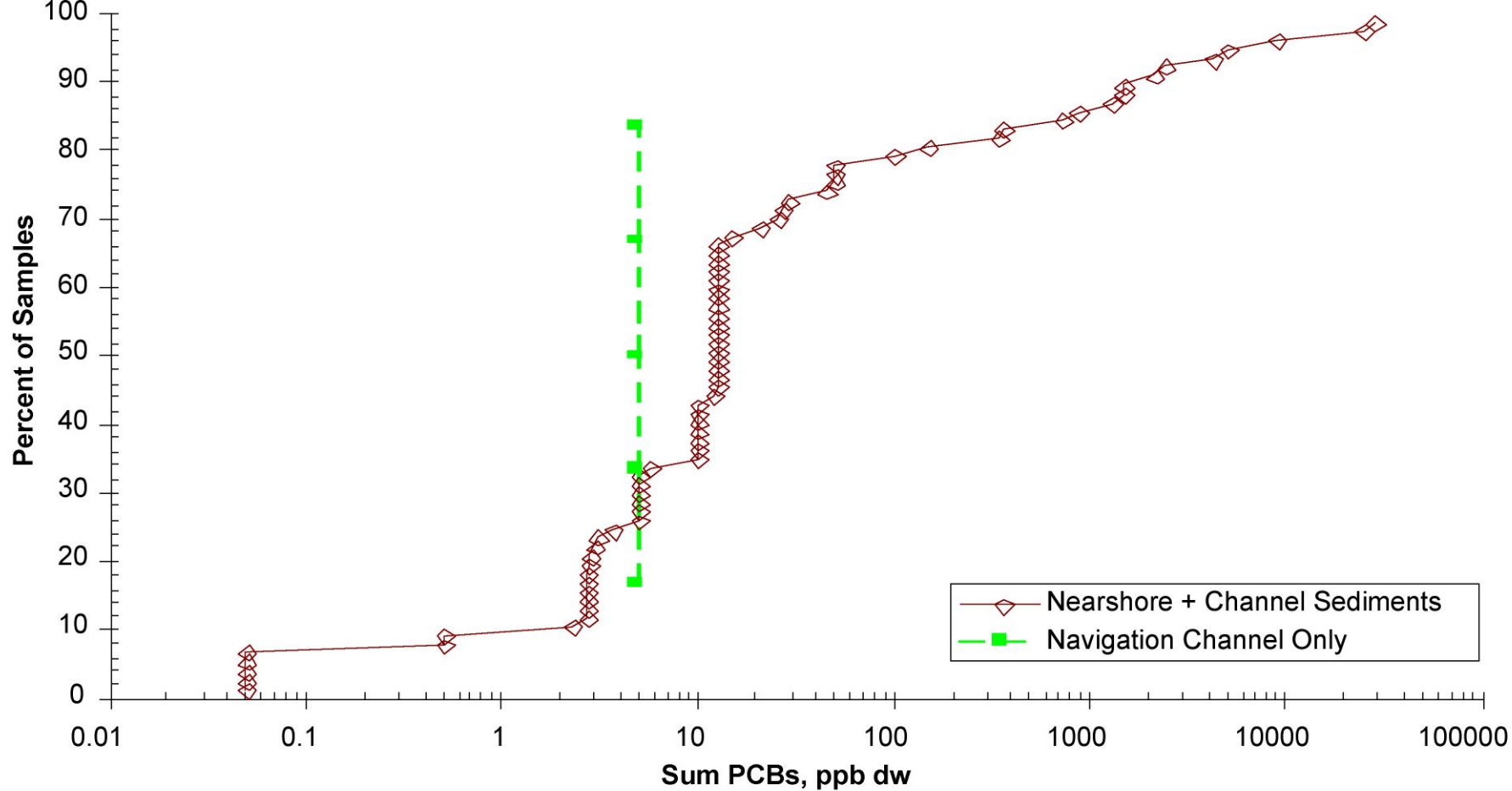
**Figure B-24**  
**Spatial Distribution of Fine Particulate**  
**Concentrations Near Willamette-**  
**Columbia Confluence**



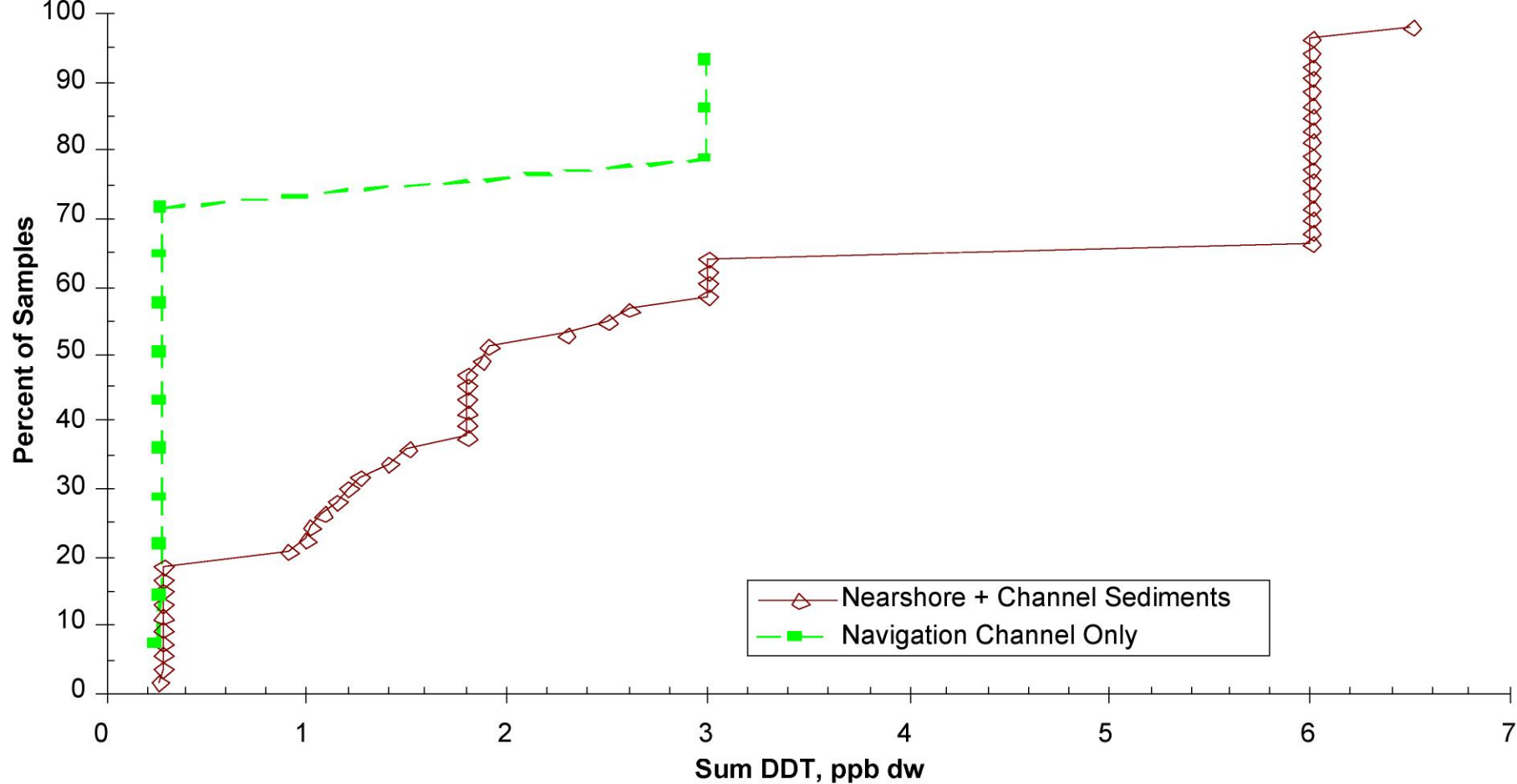
**Figure B-25**  
**Sum PCB Concentrations in Columbia**  
**River Channel Sediments Versus All**  
**Sampling Sites: River Miles 0-40**



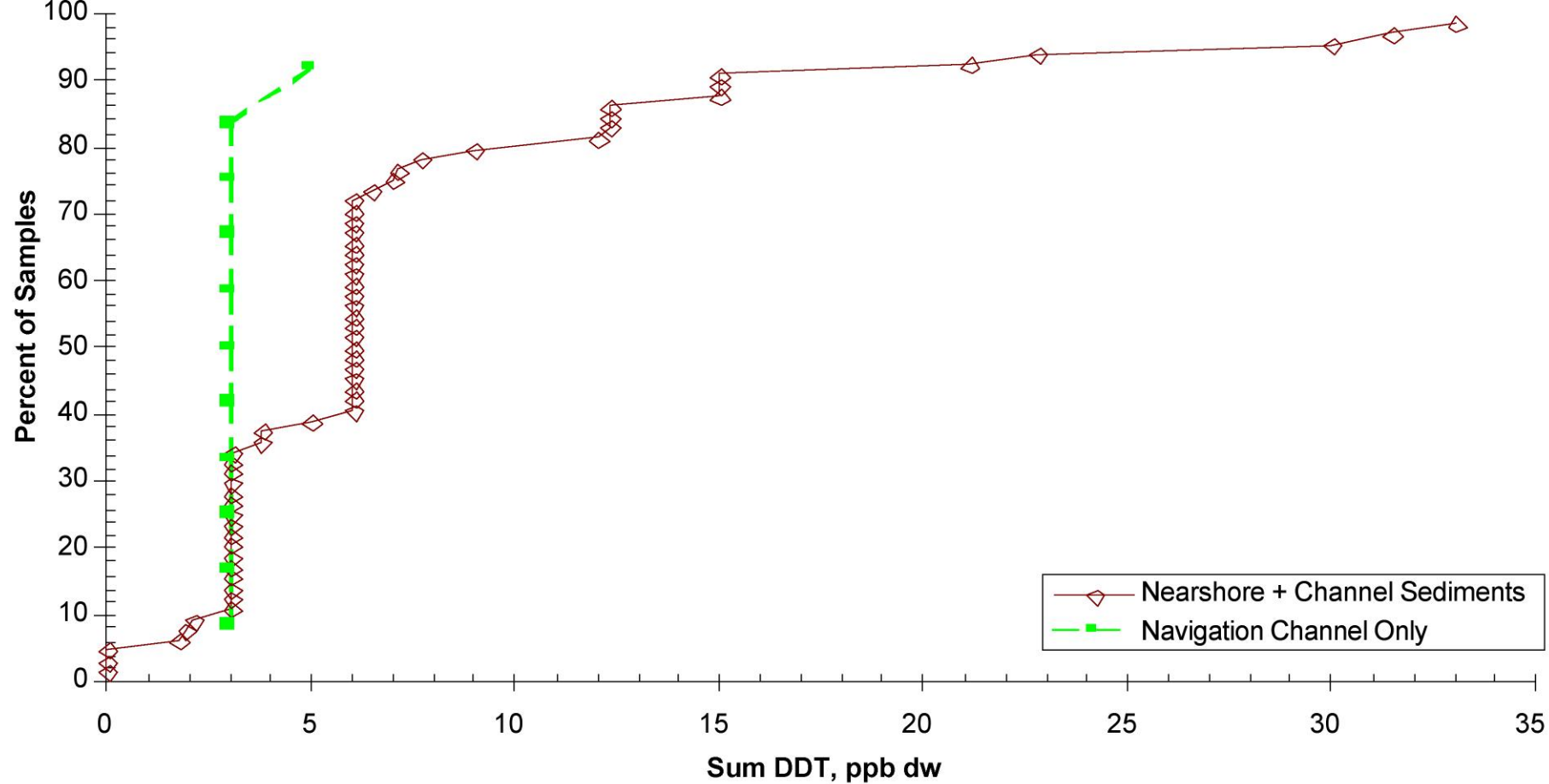
**Figure B-26**  
**Sum PCB Concentrations in Columbia**  
**River Channel Sediments Versus All**  
**Sampling Sites: River Miles 41-101**



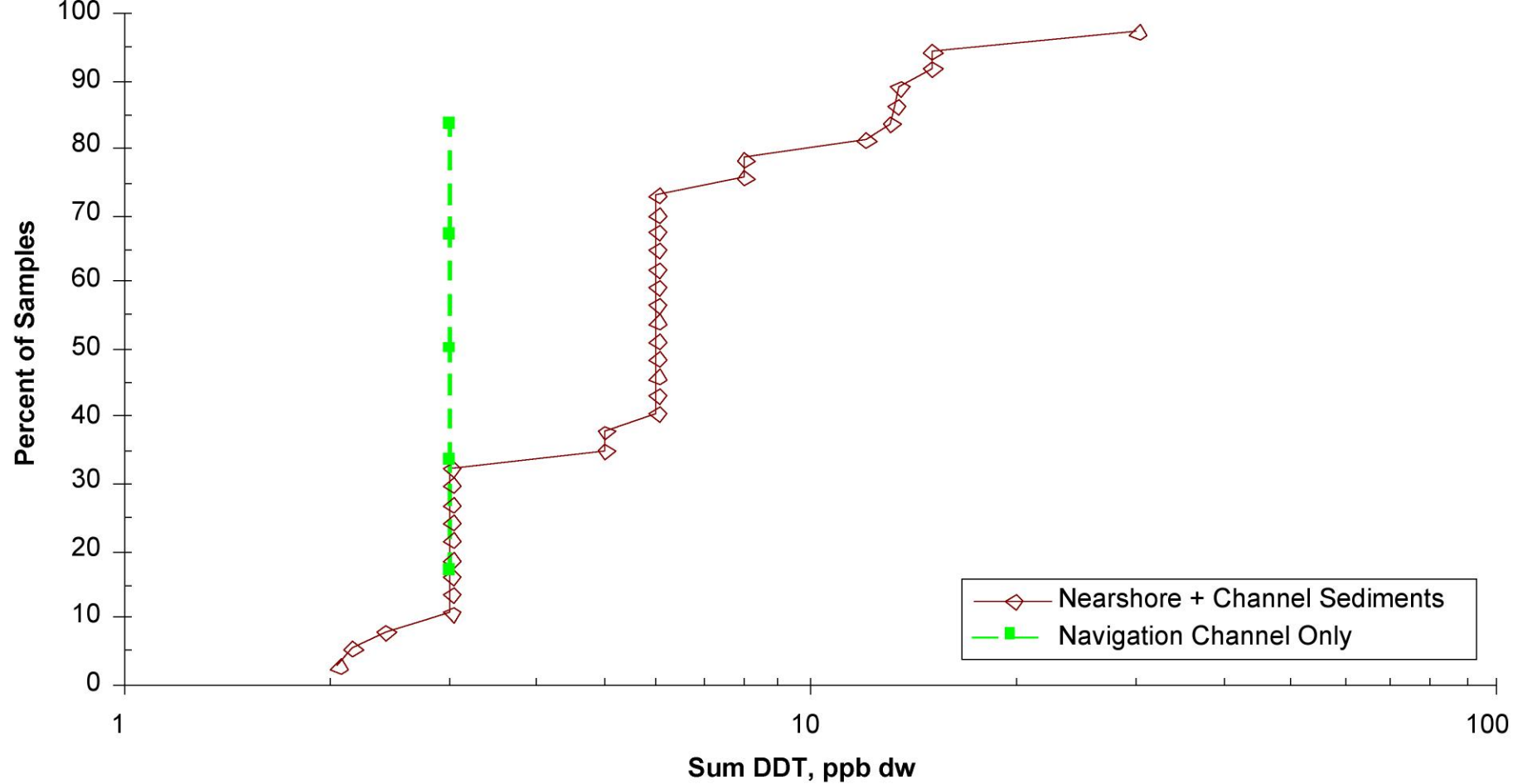
**Figure B-27**  
**Sum PCB Concentrations in Columbia**  
**River Channel Sediments Versus All**  
**Sampling Sites: Upstream of River Mile 101**



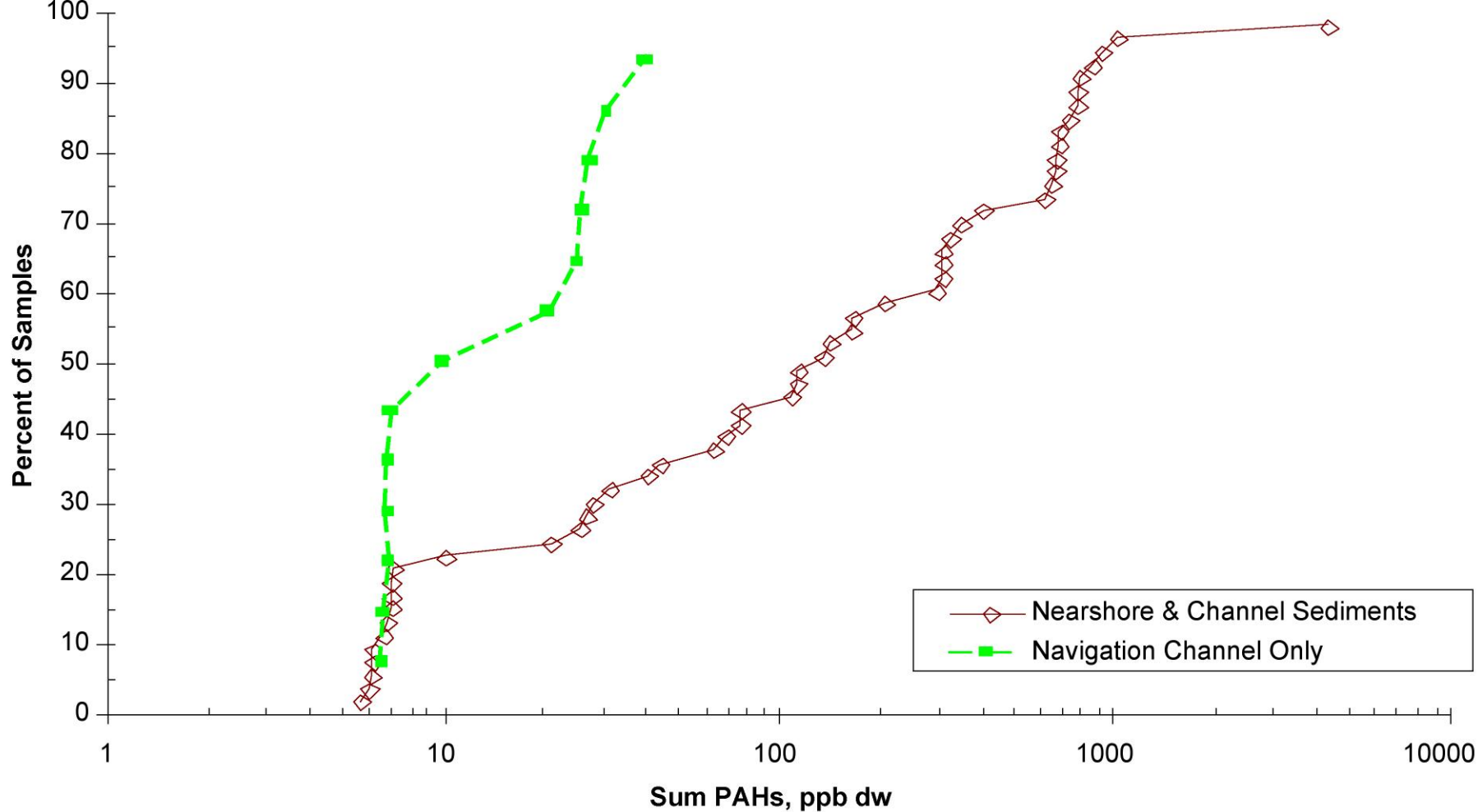
**Figure B-28**  
**Concentrations of DDT and Metabolites in**  
**Columbia River Channel Sediments Versus**  
**All Sampling Sites: River Miles 0-40**



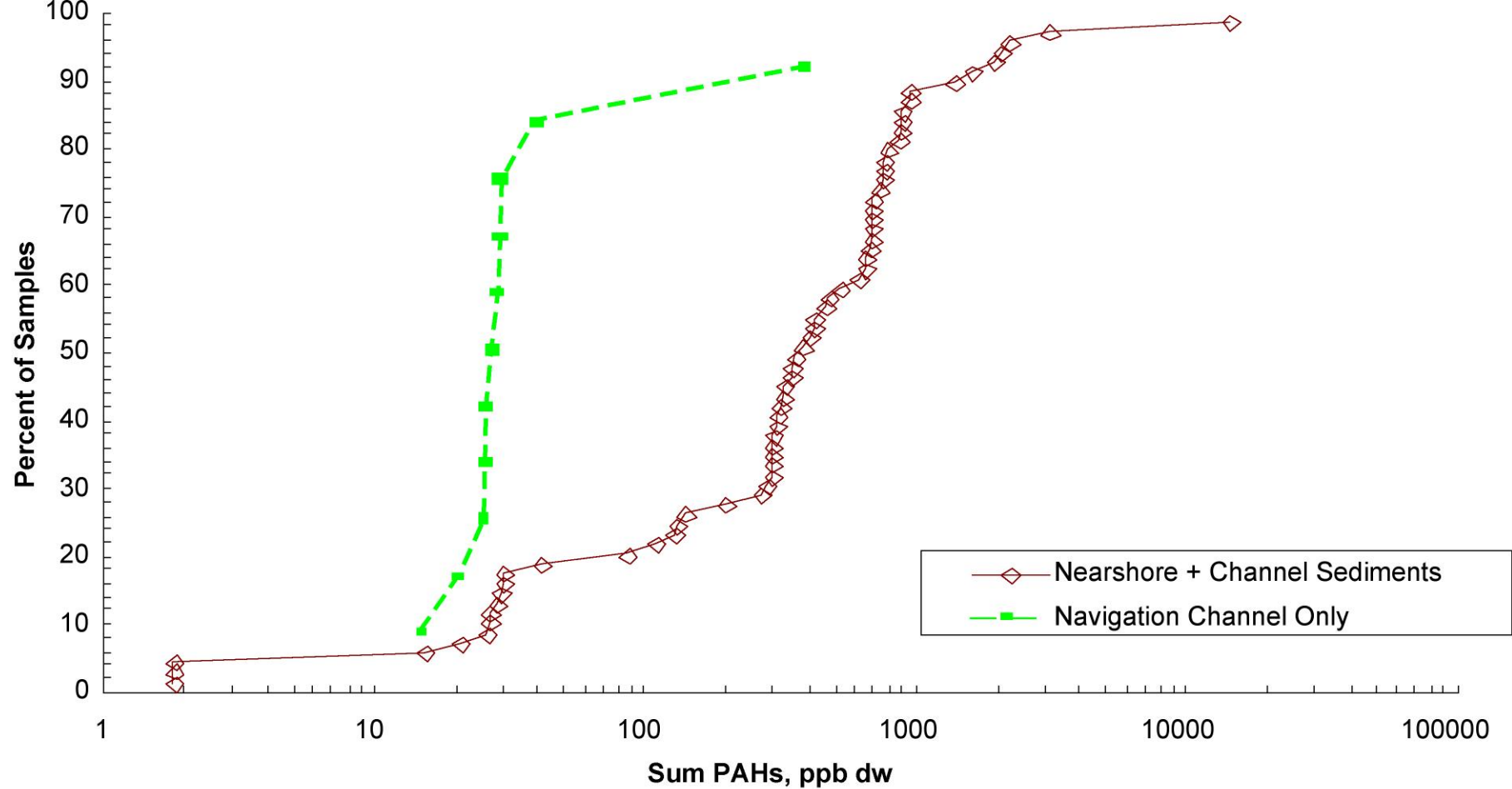
**Figure B-29**  
**Concentrations of DDT and Metabolites in**  
**Columbia River Channel Sediments Versus**  
**All Sampling Sites: River Miles 41-101**



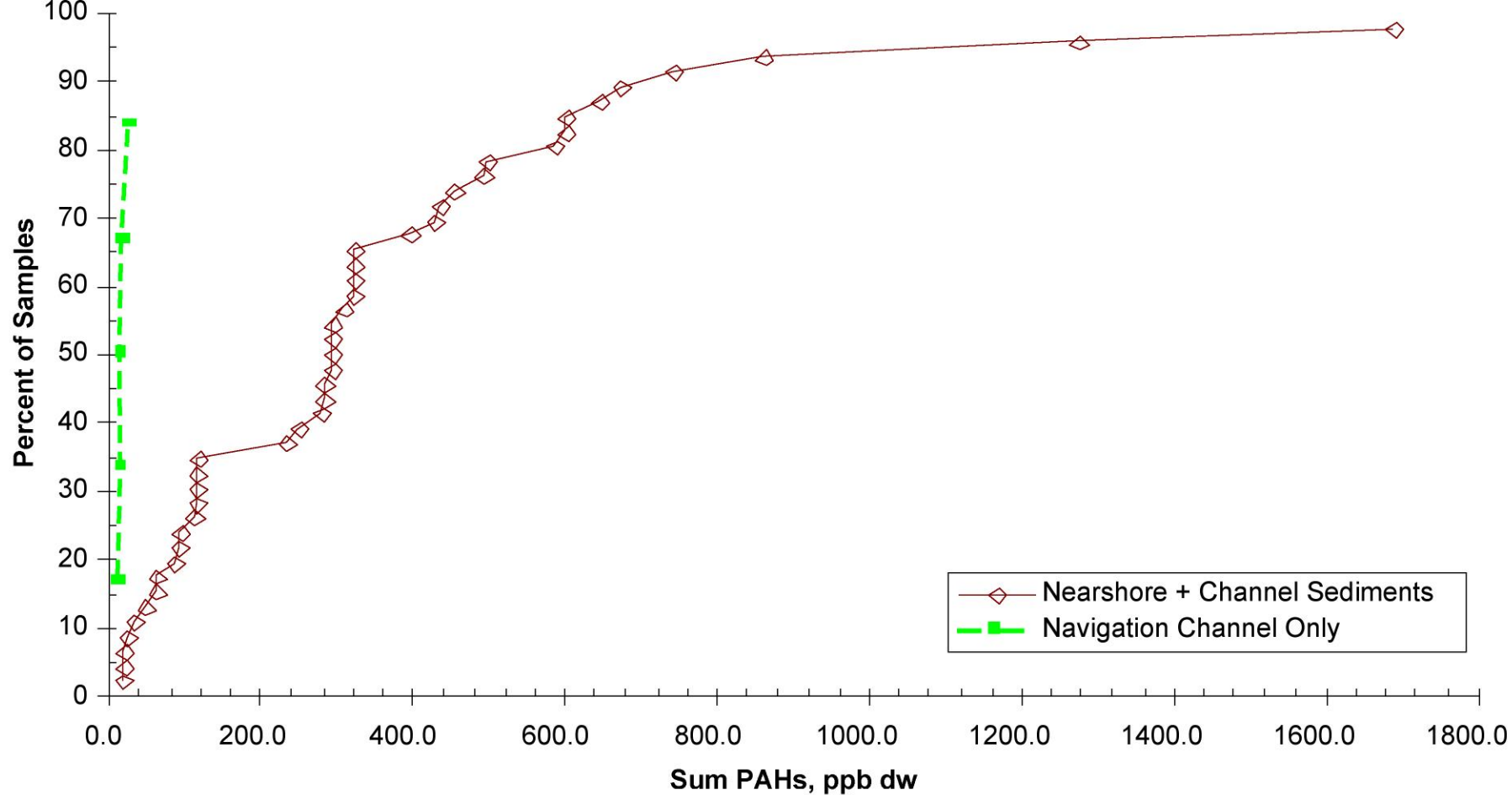
**Figure B-30**  
**Concentrations of DDT and Metabolites in**  
**Columbia River Channel Sediments Versus All**  
**Sampling Sites: Upstream of River Mile 101**



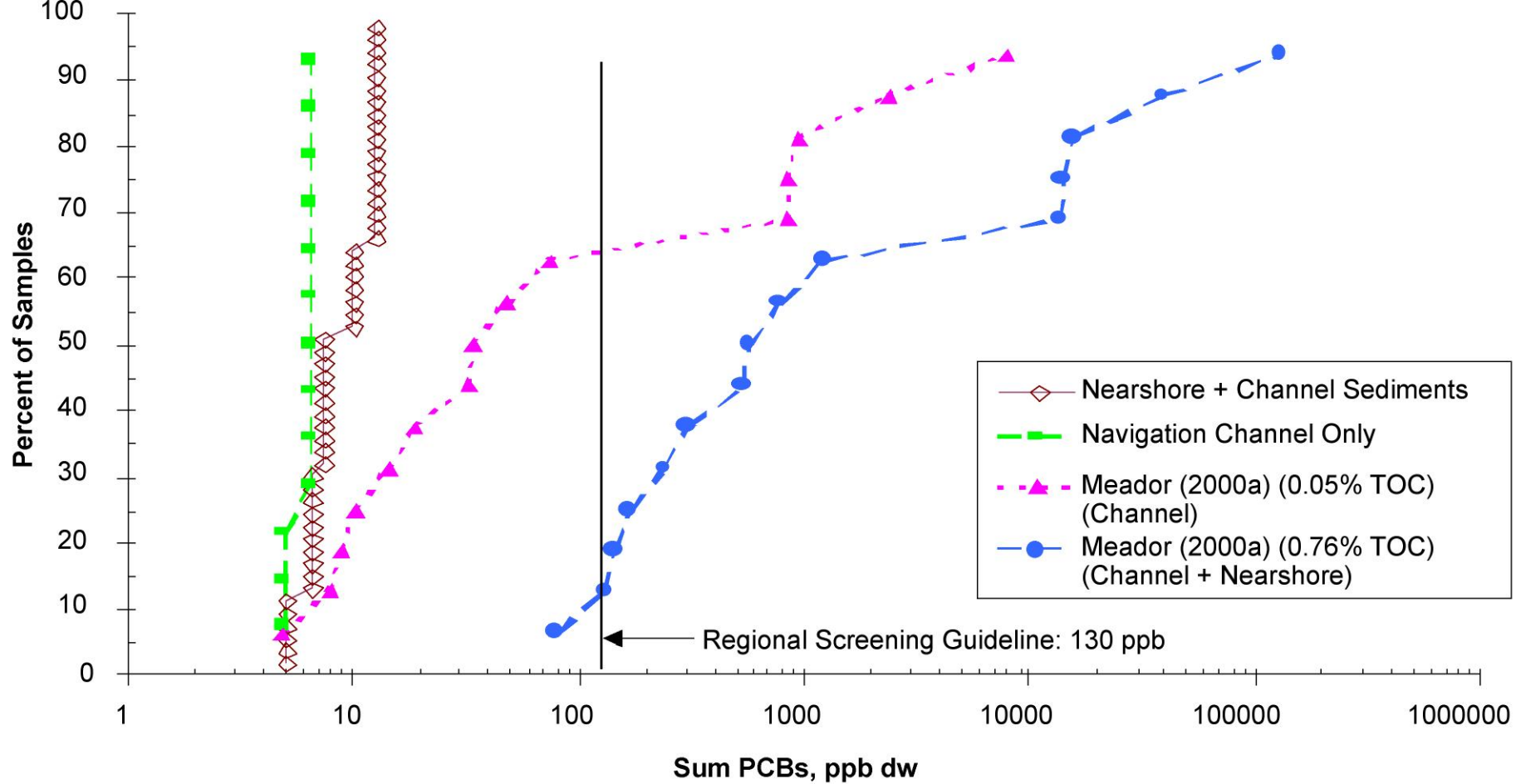
**Figure B-31**  
**Sum PAH Concentrations in Columbia River**  
**Channel Sediment Versus All Sampling Sites:**  
**River Miles 0-40**



**Figure B-32**  
**Sum PAH Concentrations in Columbia River**  
**Channel Sediment Versus All Sampling Sites:**  
**River Miles 41-101**

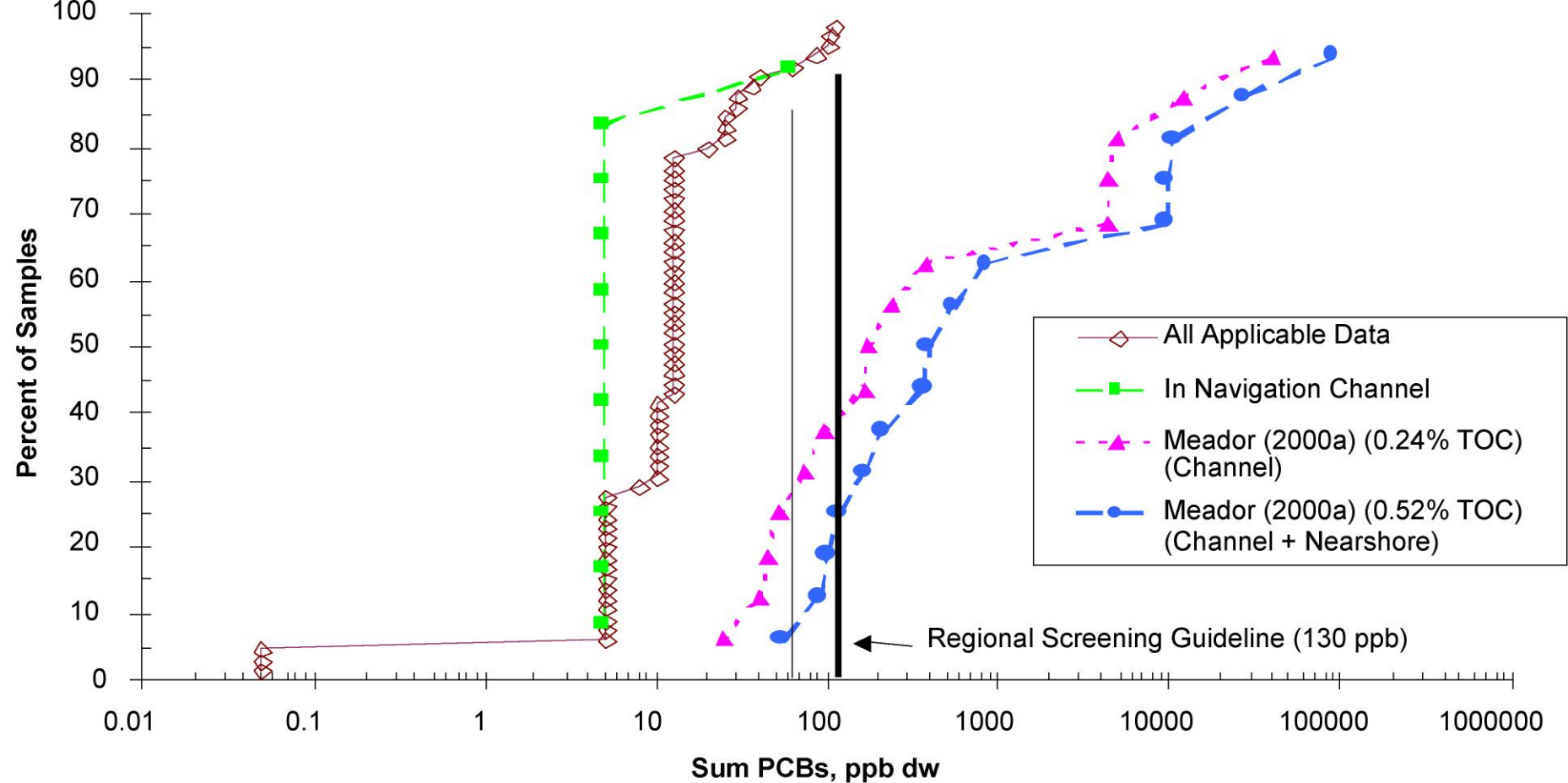


**Figure B-33**  
**Sum PAH Concentrations in Columbia River**  
**Channel Sediment Versus All Sampling Sites:**  
**Upstream of River Mile 101**



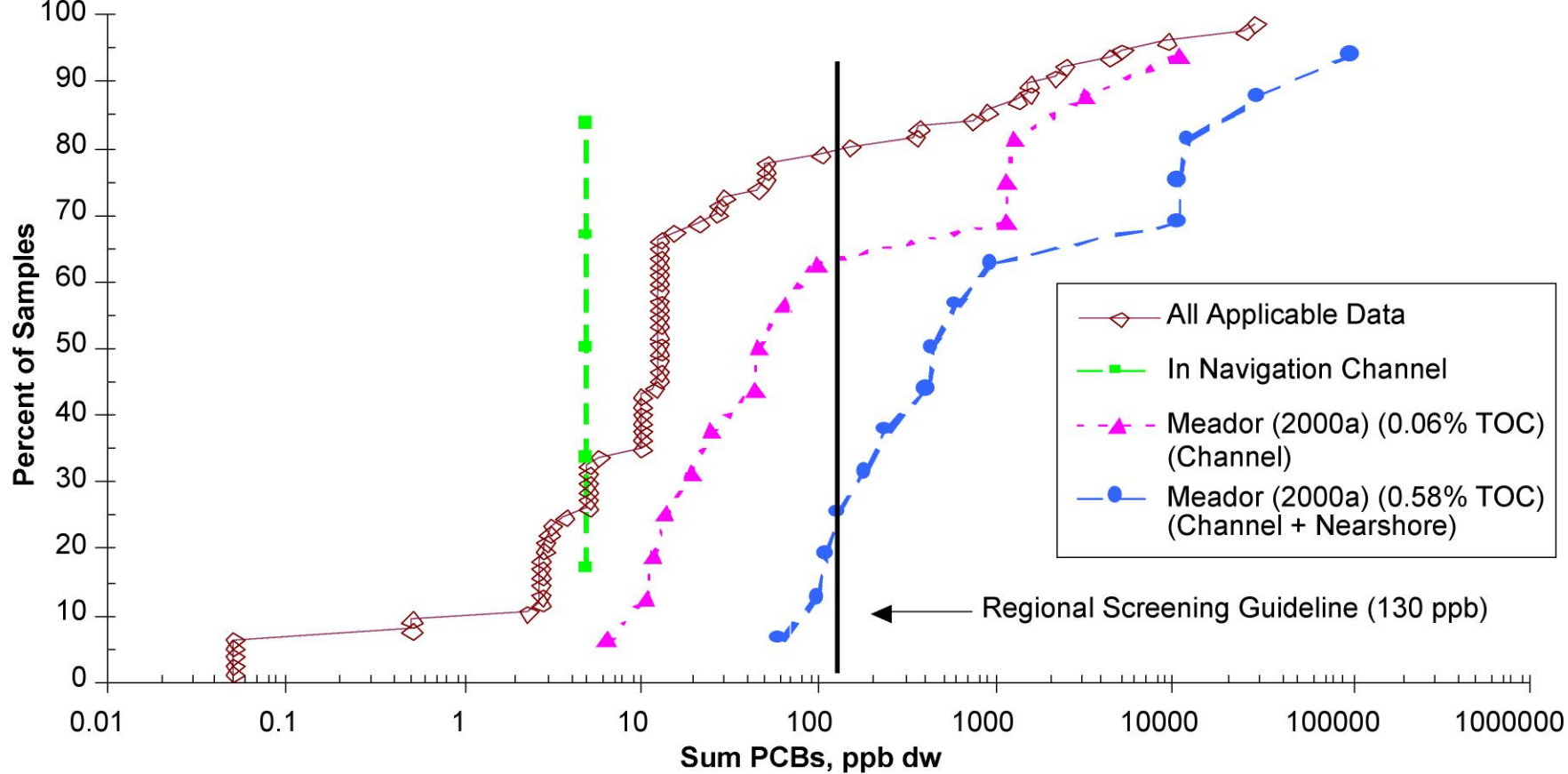
Source: Meador, 2000a

**Figure B-34**  
**Concentrations of PCBs in Sediments**  
**Compared to Those Associated with Adverse**  
**Effects in Fish: River Miles 0-40**



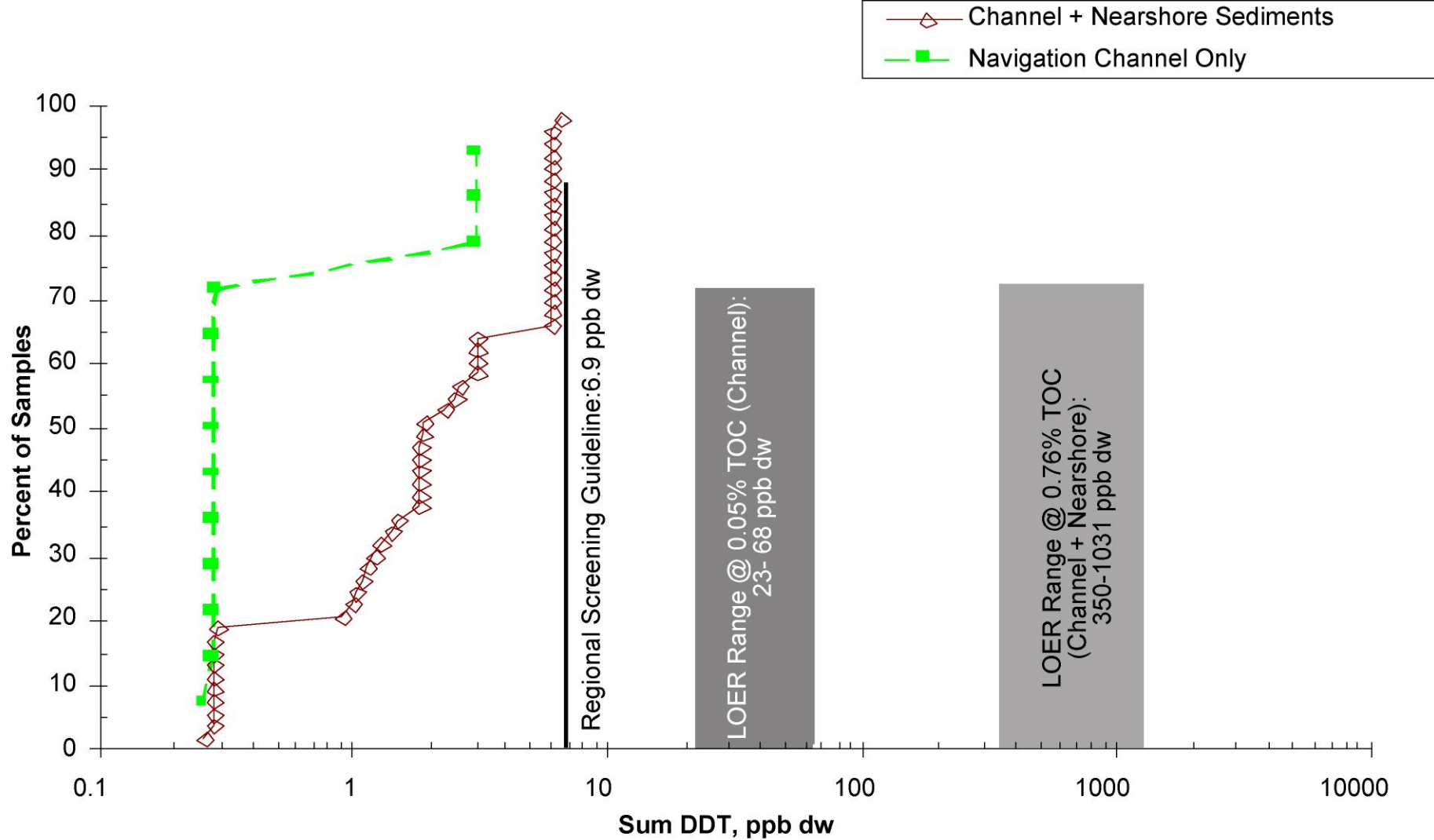
Source: Meador, 2000a

**Figure B-35**  
**Concentrations of PCBs in Sediments**  
**Compared to Those Associated with Adverse**  
**Effects in Fish: River Miles 41-101**

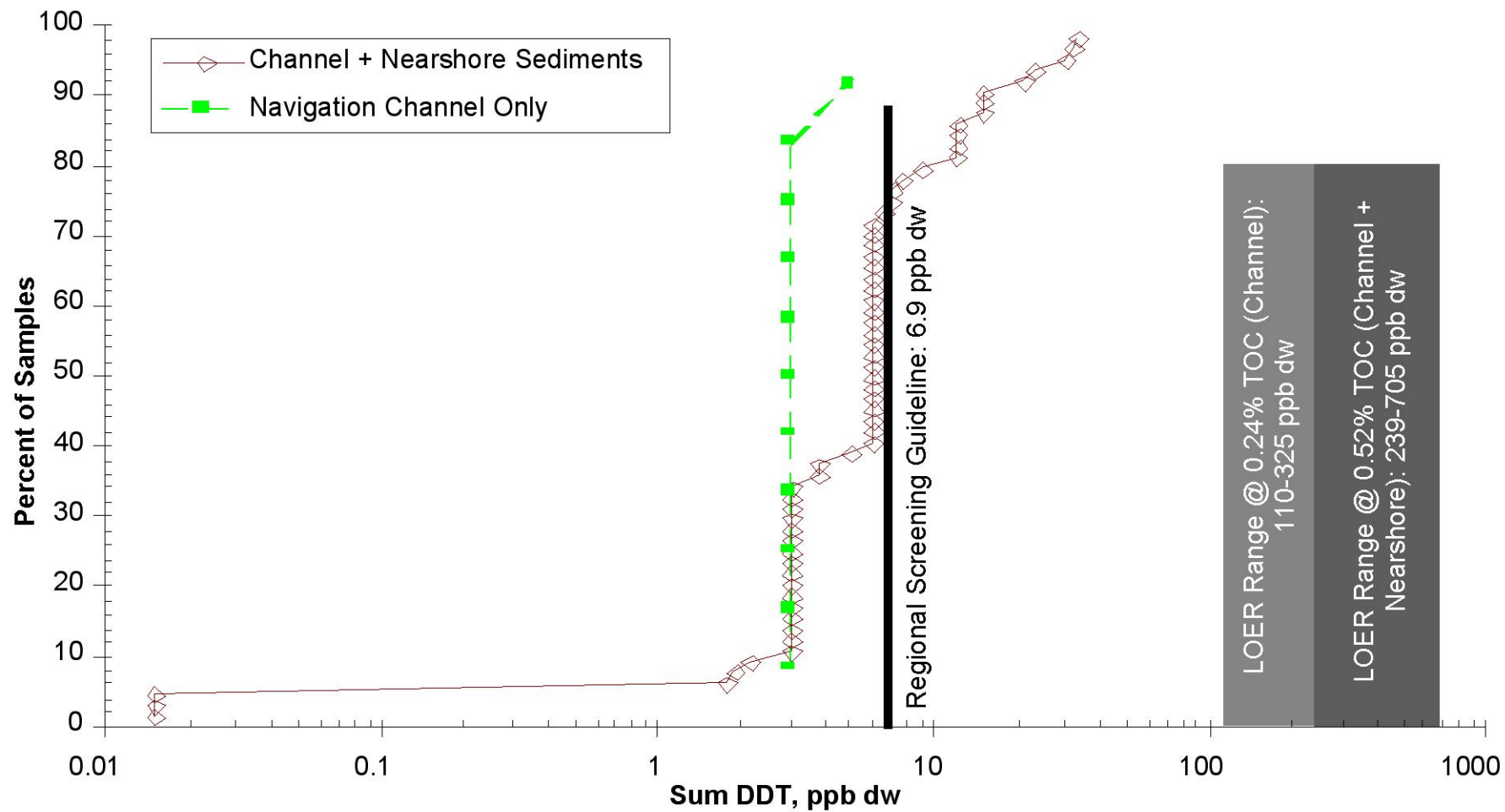


Source: Meador, 2000a

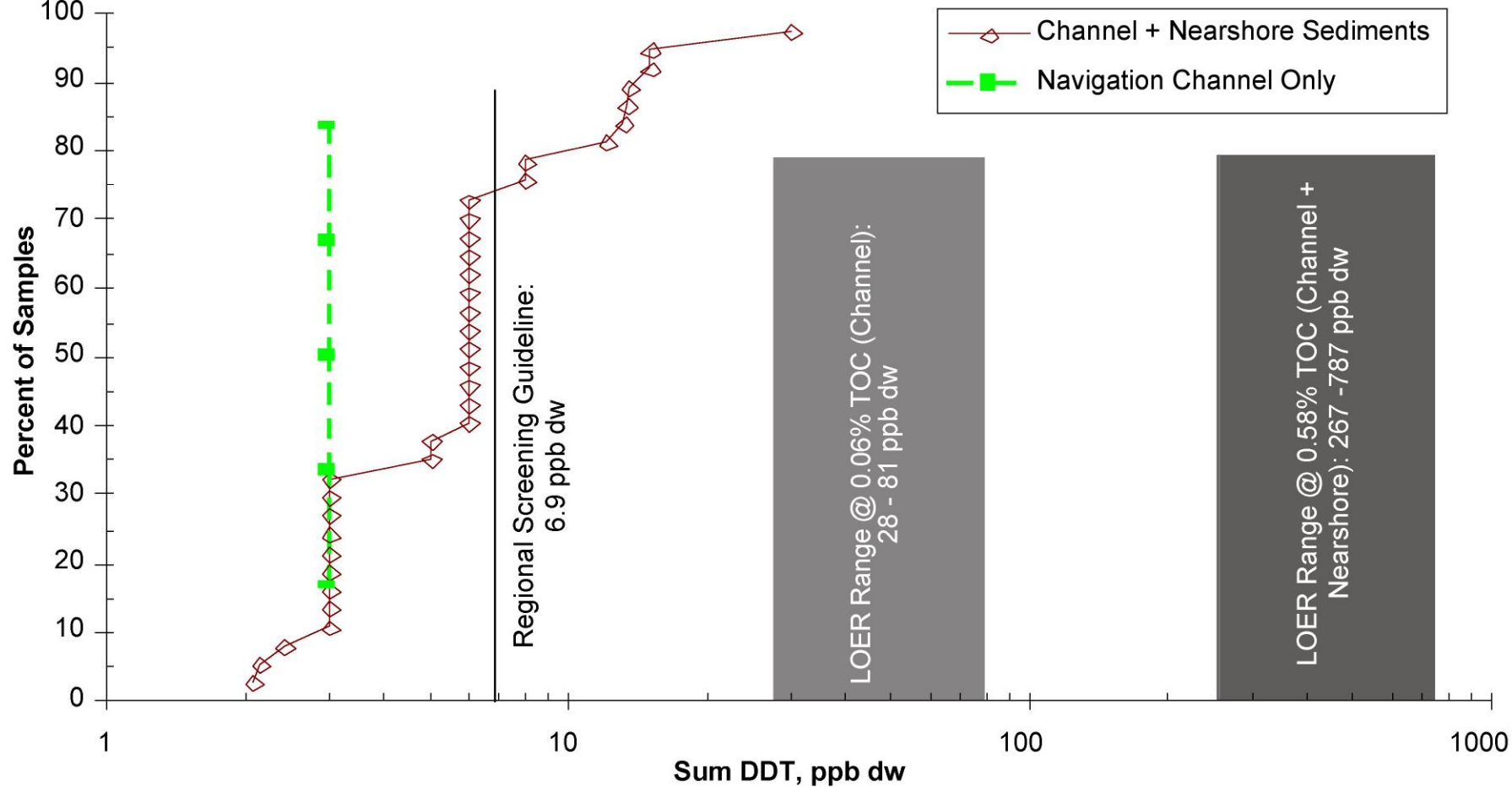
**Figure B-36**  
**Concentrations of PCBs in Sediments**  
**Compared to Those Associated with Adverse**  
**Effects in Fish: Upstream of River Mile 101**



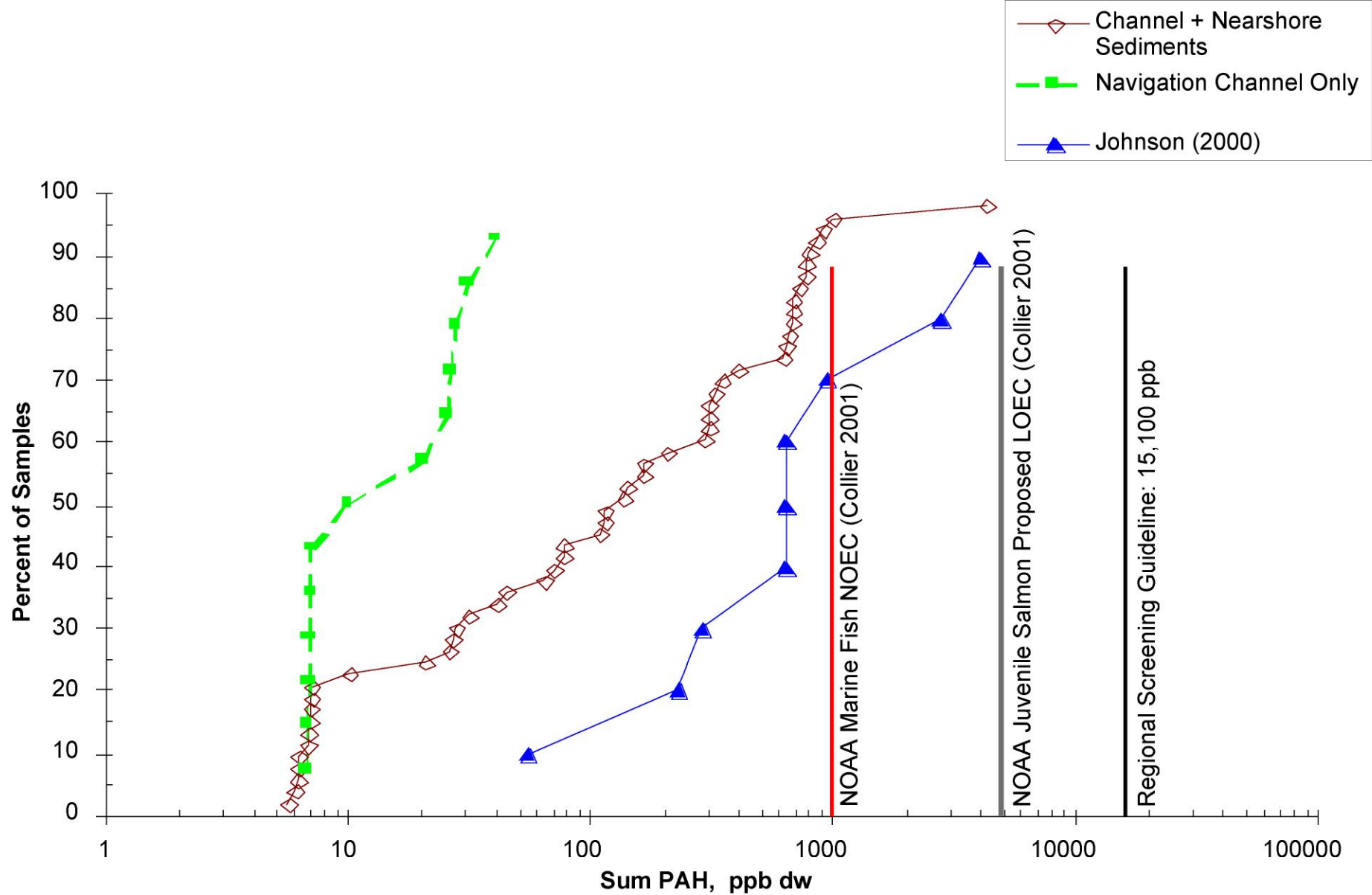
**Figure B-37**  
**Concentrations of DDT and Metabolites in**  
**Sediments Compared to Three Effects Criteria:**  
**River Miles 0-40**



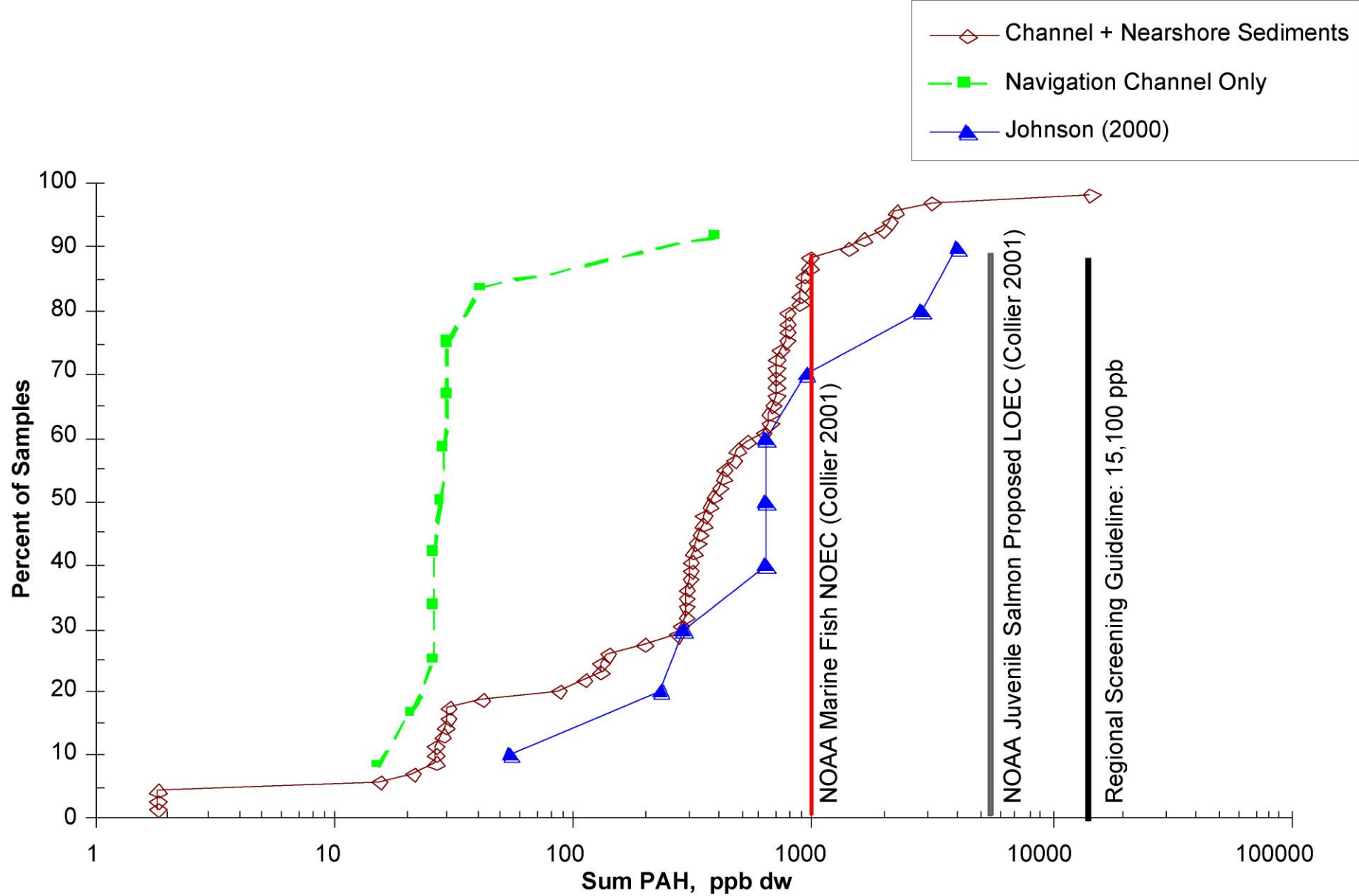
**Figure B-38**  
**Concentrations of DDT and Metabolites in**  
**Sediments Compared to Three Effects Criteria:**  
**River Miles 41-101**



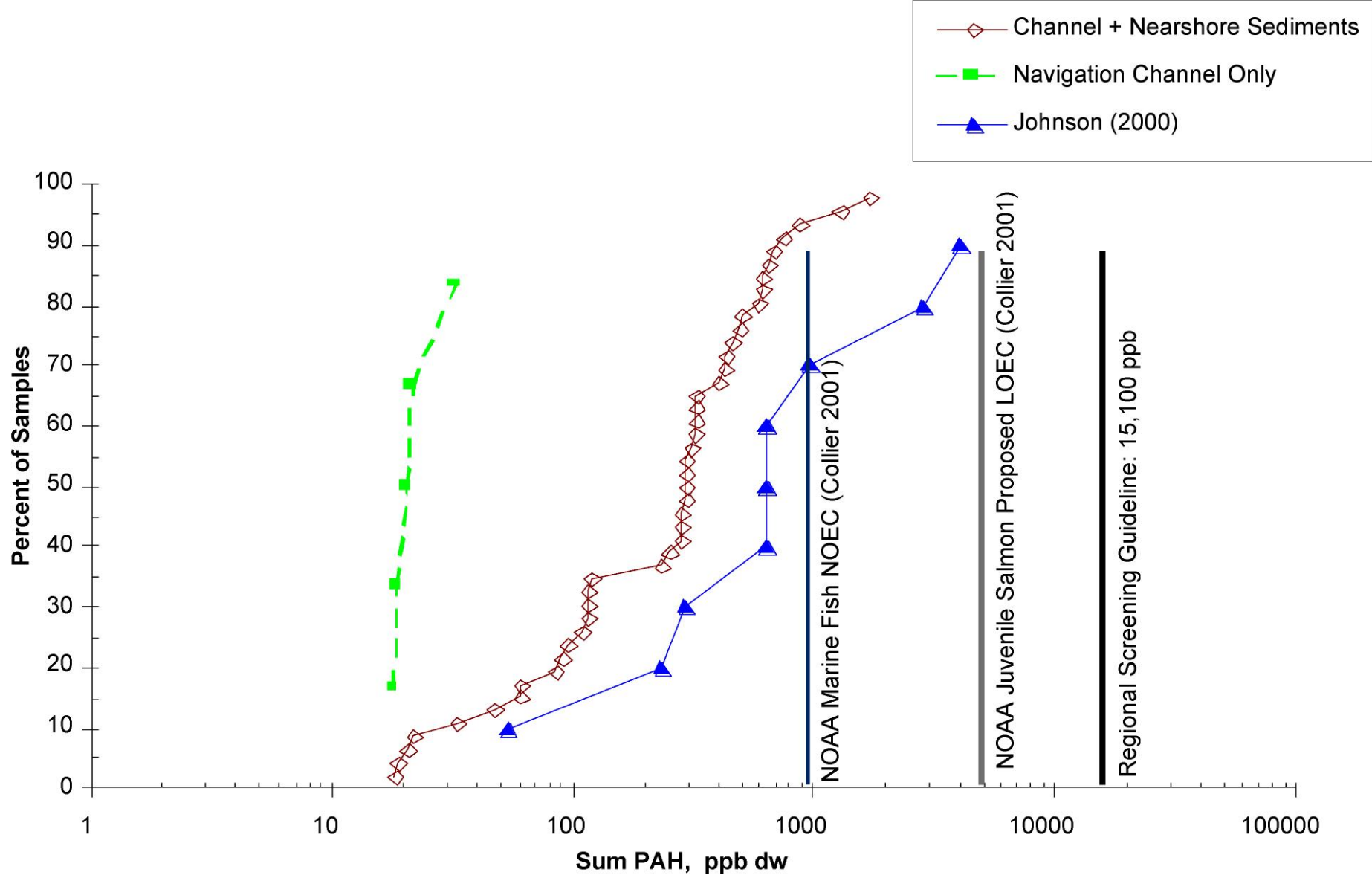
**Figure B-39**  
**Concentrations of DDT and Metabolites in**  
**Sediments Compared to Three Effects Criteria:**  
**Upstream of River Mile 101**



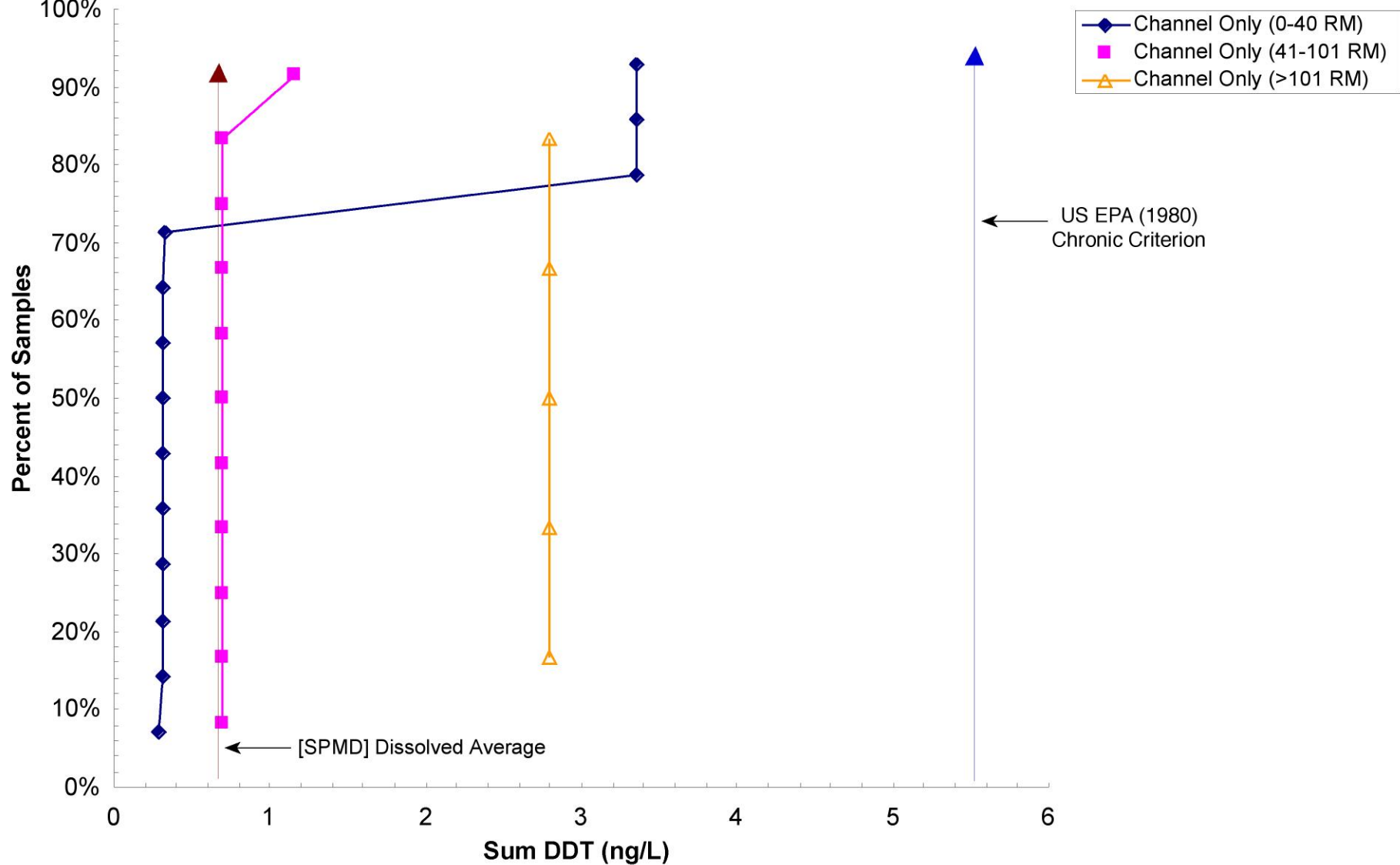
**Figure B-40**  
**Concentrations of PAHs in Sediments Compared**  
**to Four Effects Criteria: River Miles 0-40**



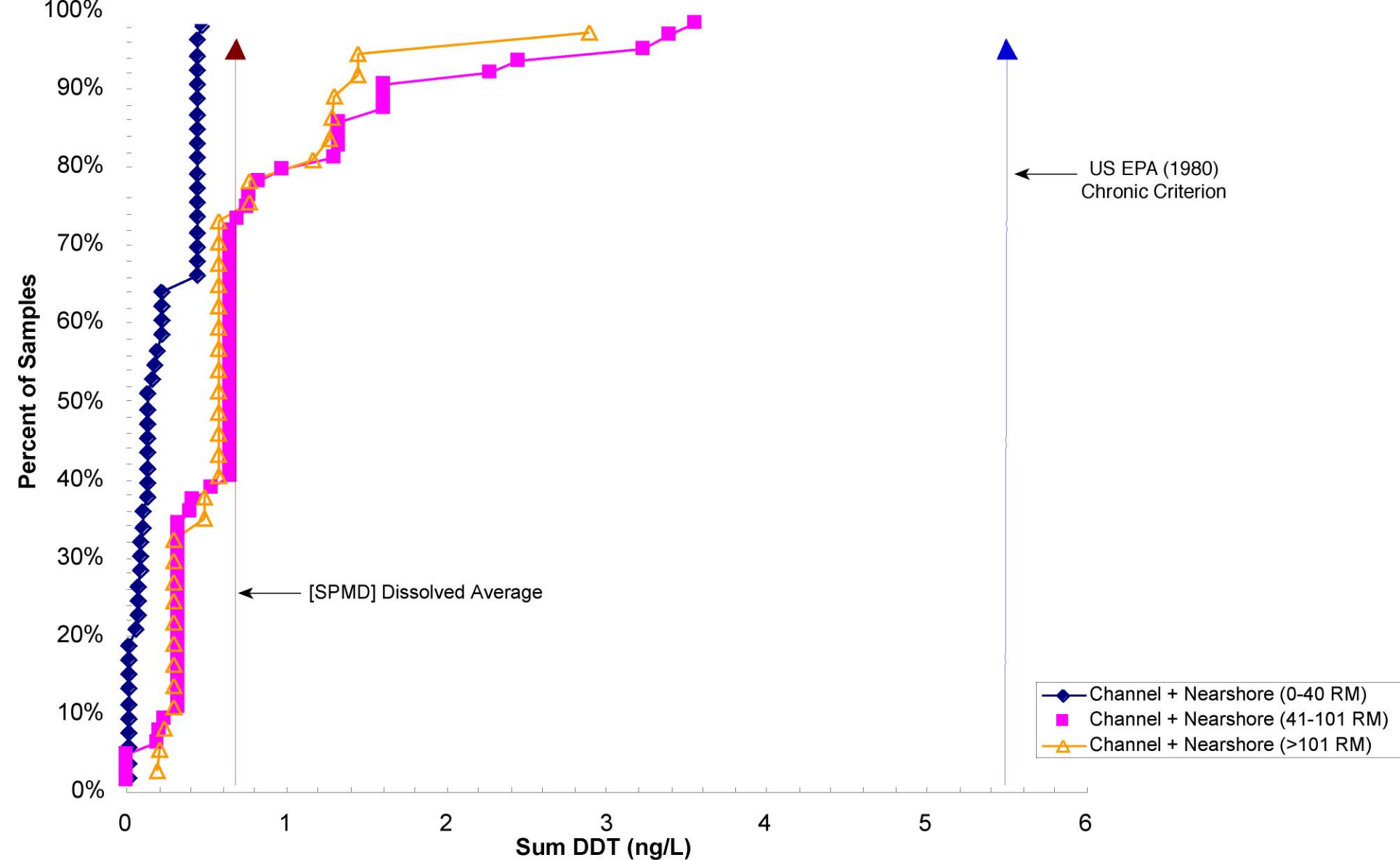
**Figure B-41**  
**Concentrations of PAHs in Sediments Compared**  
**to Four Effects Criteria: River Miles 41-101**



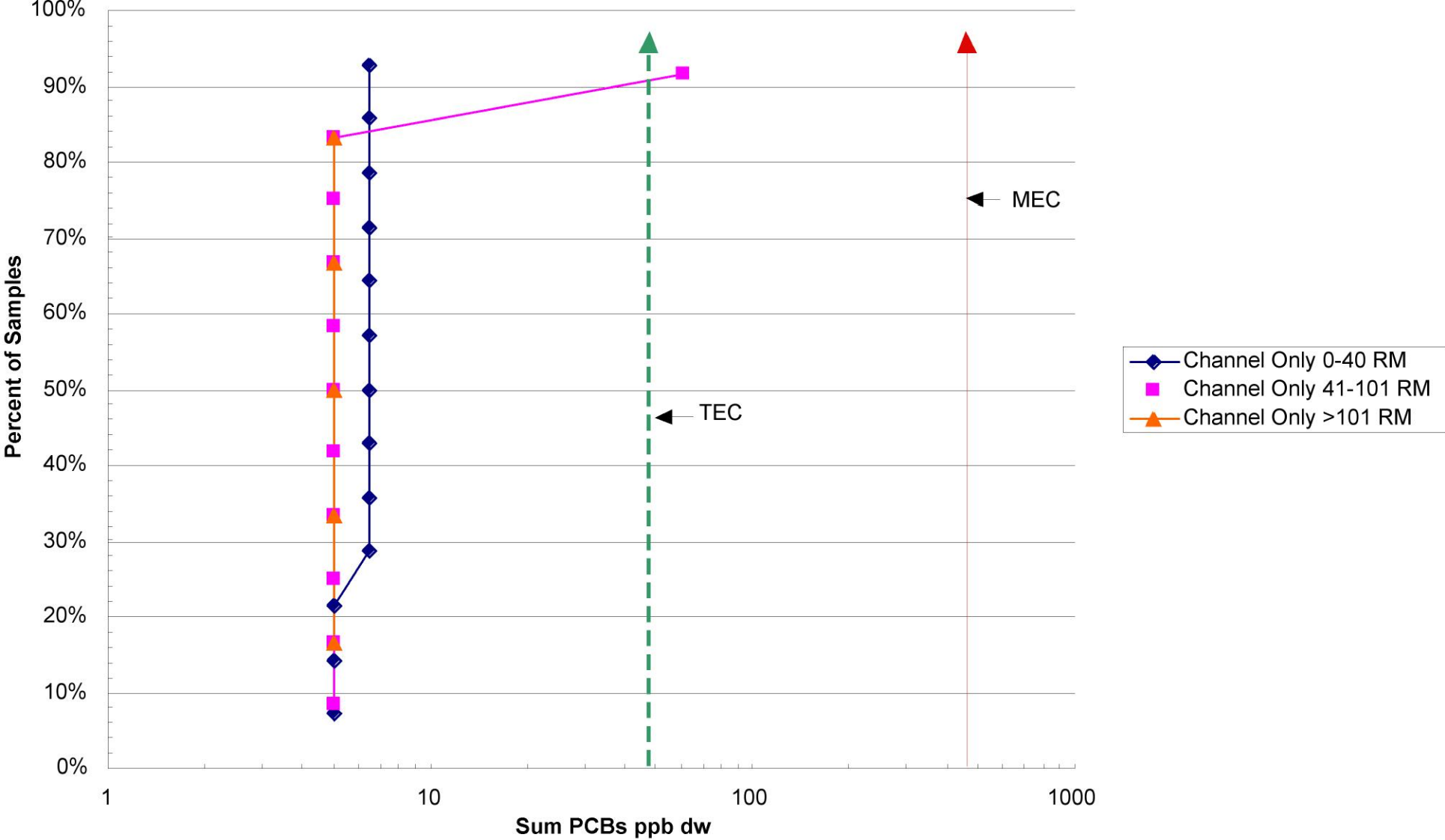
**Figure B-42**  
**Concentrations of PAHs in Sediments Compared to**  
**Four Effects Criteria: Upstream of River Mile 101**



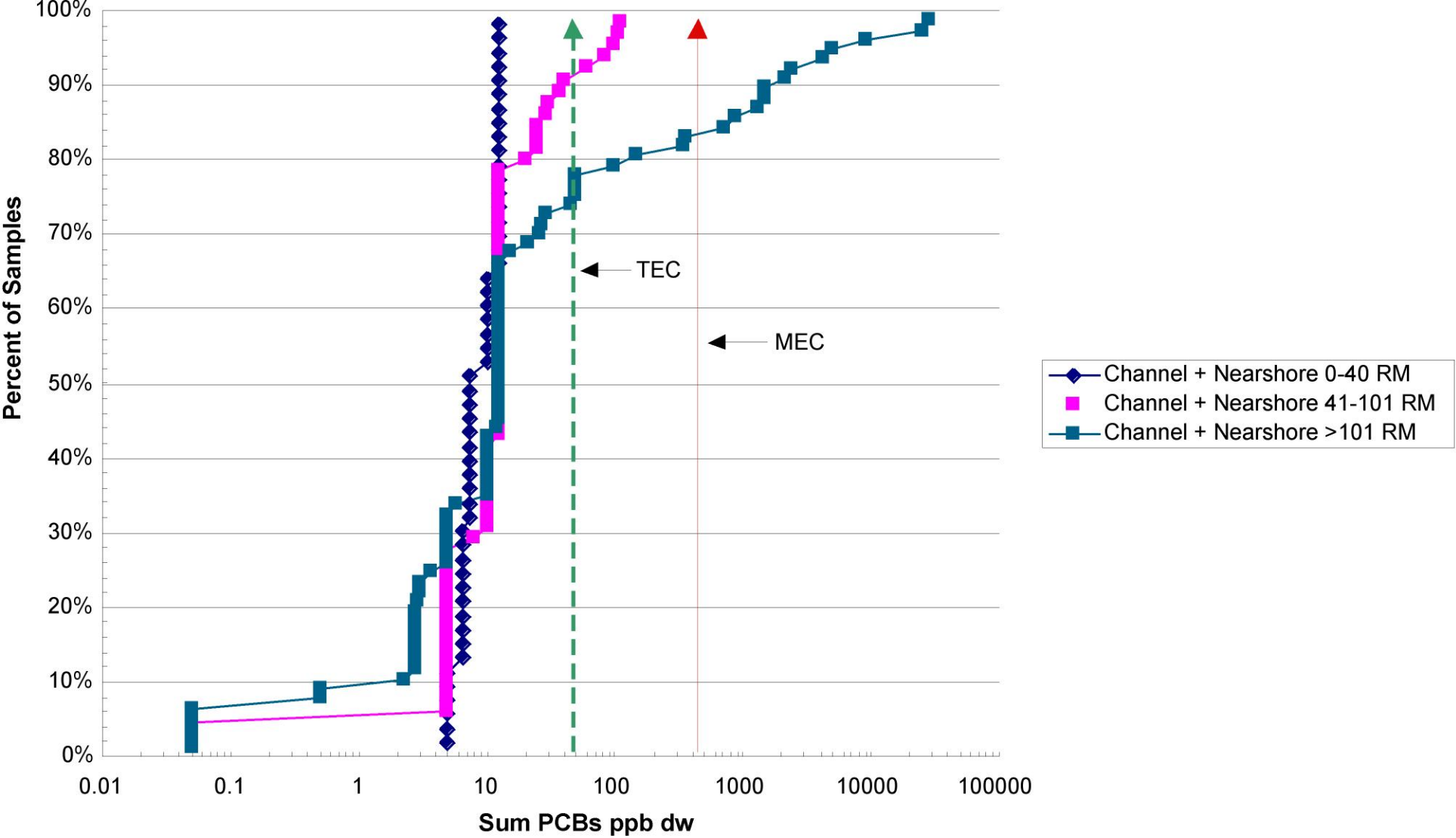
**Figure B-43**  
**Estimated Aqueous Concentrations of DDT and Metabolites in**  
**Sediment Porewaters Compared to Those Associated with**  
**Adverse Effects to Aquatic Organisms: Channel Sediments**



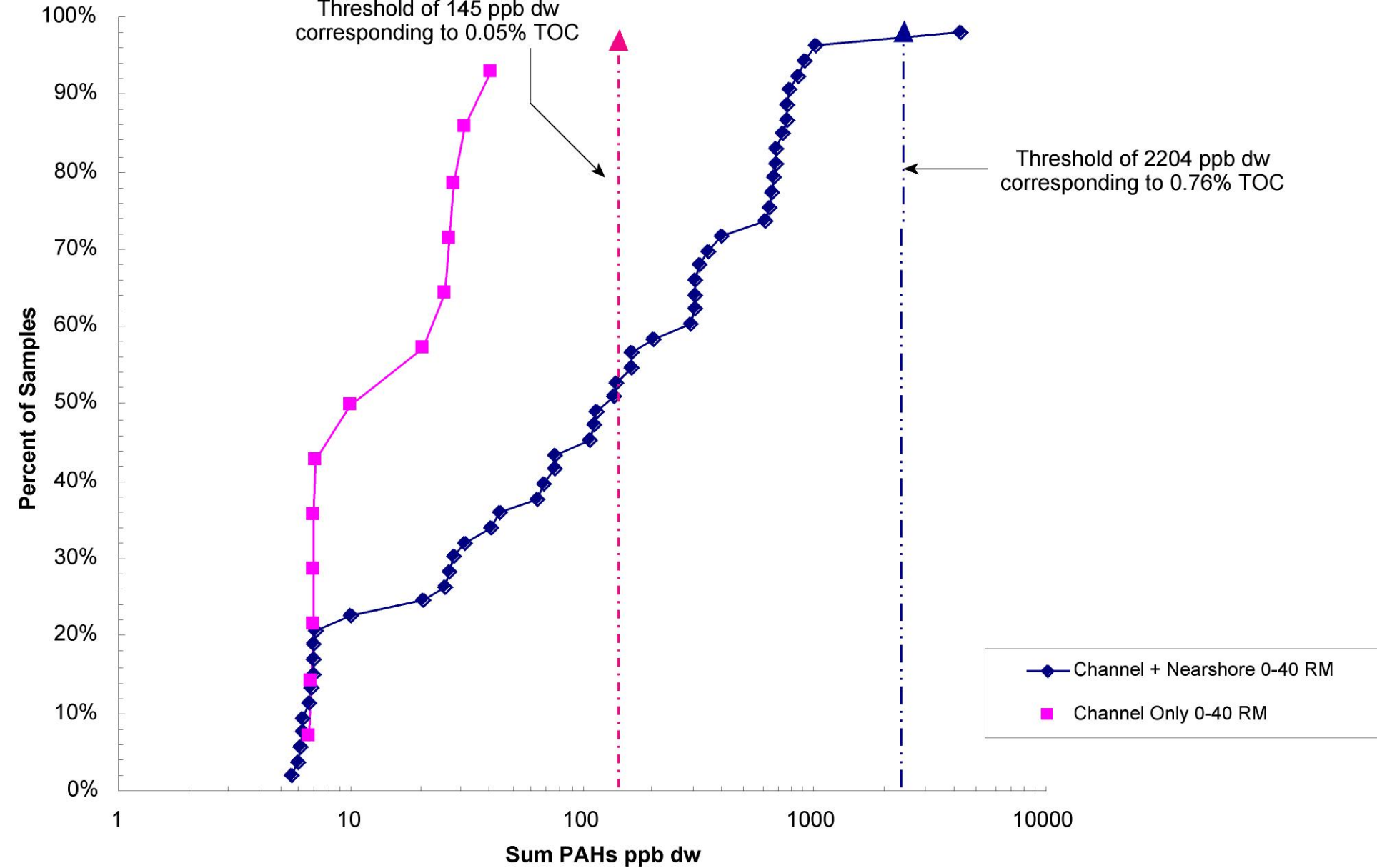
**Figure B-44**  
**Estimated Aqueous Concentrations of DDT and Metabolites in Sediment Porewaters Compared to Those Associated with Adverse Effects to Aquatic Organisms: Channel and Nearshore Sediments Combined**



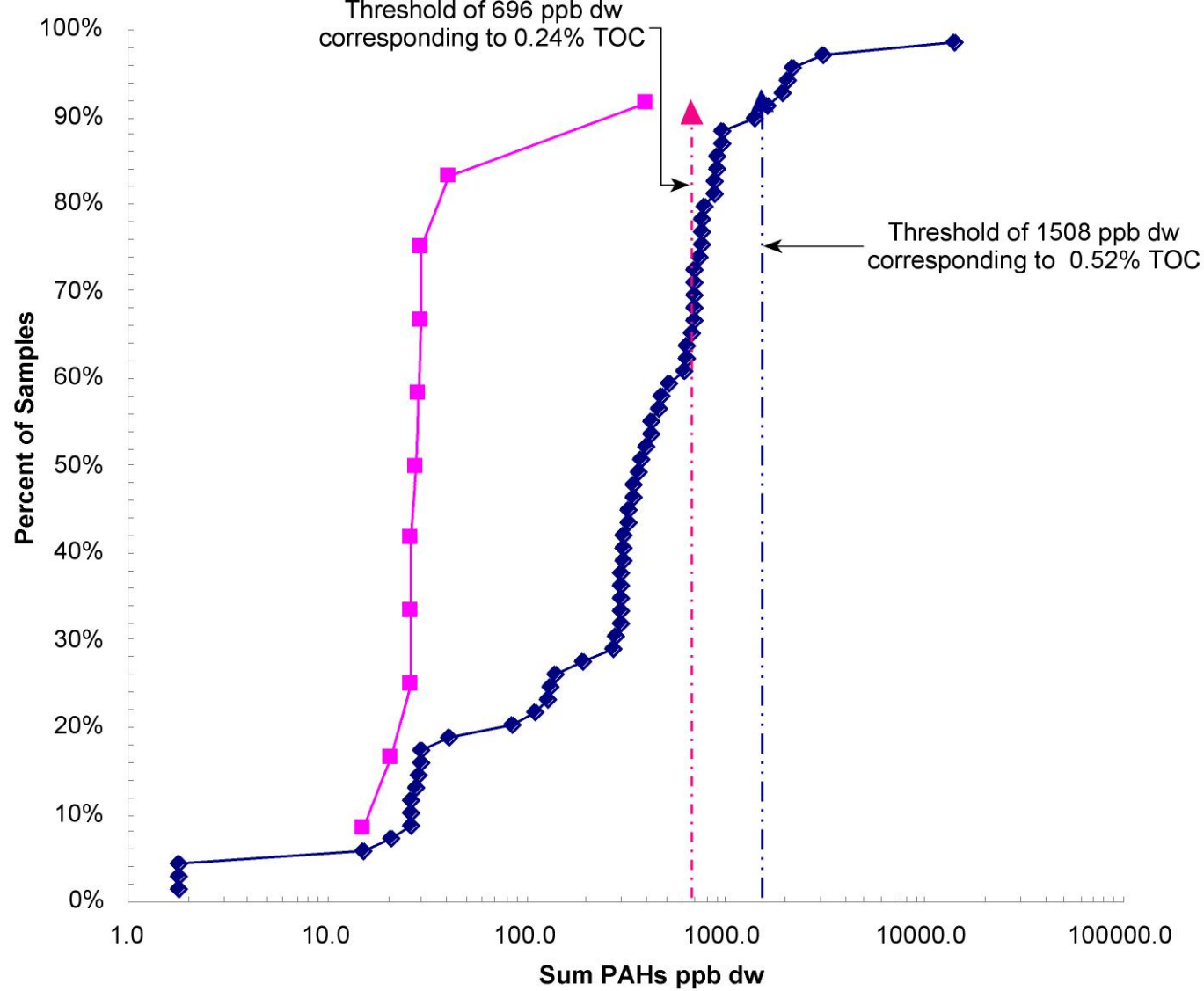
**Figure B-45**  
**Estimated Dry Weight Concentrations of PCBs in Sediment**  
**Compared to Those Associated with Adverse Effects to Aquatic**  
**Organisms: Channel Sediments**



**Figure B-46**  
**Estimated Dry Weight Concentrations of PCBs in Sediment**  
**Compared to Those Associated with Adverse Effects to Aquatic**  
**Organisms: Channel and Nearshore Sediments Combined**

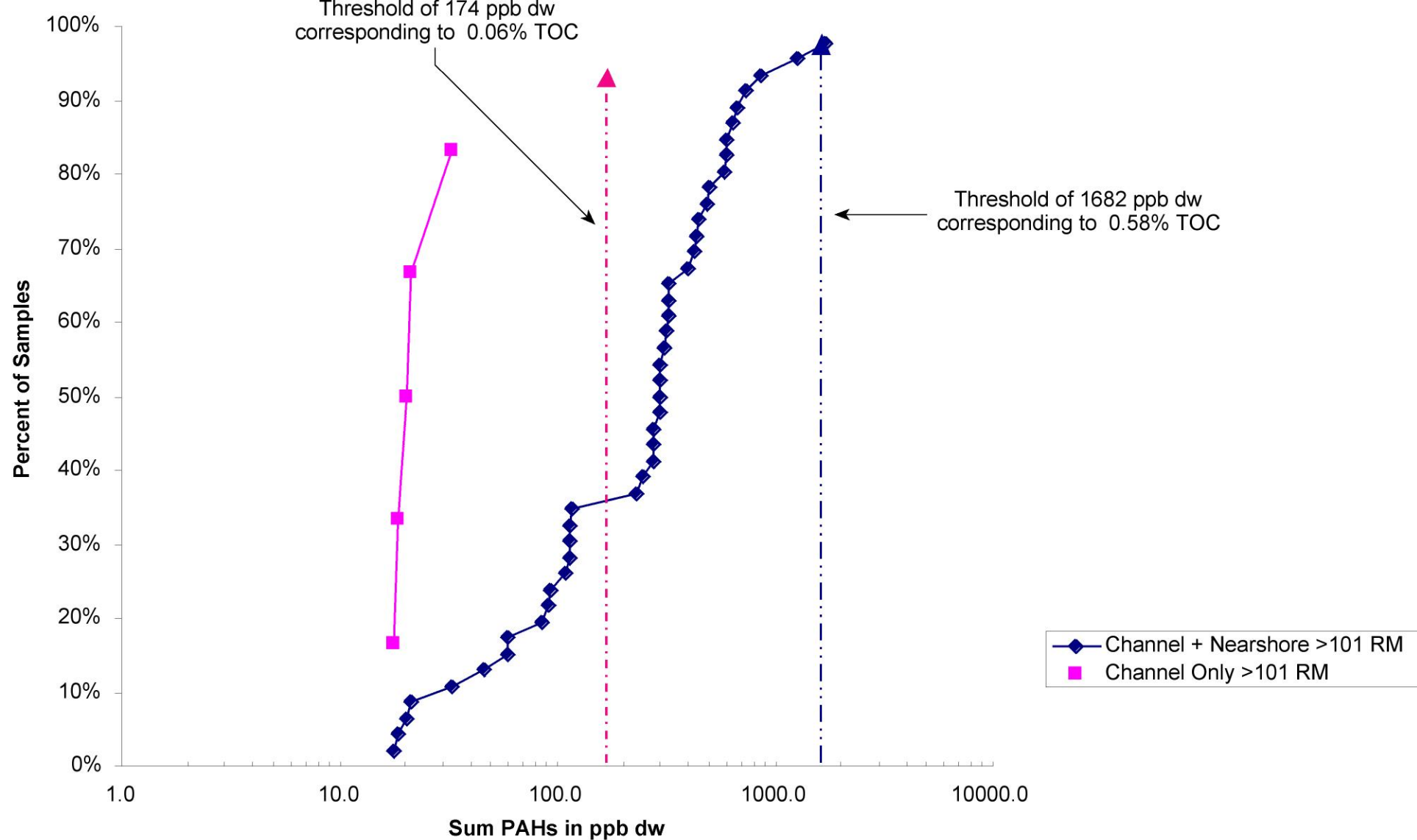


**Figure B-47**  
**Estimated Dry Weight Concentrations of PAHs in Sediment**  
**Compared to Those Associated with Adverse Effects to Aquatic**  
**Organisms: River Mile 0-40**



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**Figure B-48**  
**Estimated Dry Weight Concentrations of PAHs in Sediment**  
**Compared to Those Associated with Adverse Effects to Aquatic**  
**Organisms: River Mile 41-101**



**Figure B-49**  
**Estimated Dry Weight Concentrations of PAHs in Sediment**  
**Compared to Those Associated with Adverse Effects to Aquatic**  
**Organisms: River Mile >101**